Russian River Biological Opinion Status and Data Report

Year 2015 - 2016

Sonoma Water

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Accessibility Statement

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APPENDICES

CHAPTER 1 : Introduction

On September 24, 2008, the National Marine Fisheries Service (NMFS) issued a 15-year Biological Opinion for water supply, flood control operations, and channel maintenance conducted by the U.S. Army Corps of Engineers (USACE), Sonoma County Water Agency (Water Agency), and Mendocino County Russian River Flood Control and Water Conservation Improvement District in the Russian River watershed (NMFS 2008). The Biological Opinion authorizes incidental take of threatened and endangered Chinook salmon, coho salmon, and steelhead pending implementation of a Reasonable and Prudent Alternative (RPA) to status quo management of reservoir releases, river flow, habitat condition, and facilities in portions of the mainstem Russian River, Dry Creek, and Russian River Estuary. Mandated projects to ameliorate impacts to listed salmonids in the RPA are partitioned among USACE and the Water Agency. Each organization has its own reporting requirements to NMFS. Because coho salmon are also listed as endangered by the California Endangered Species Act (CESA), the Water Agency is party to a Consistency Determination issued by the California Department of Fish and Wildlife (CDFW) in November 2009. The Consistency Determination mandates that the Water Agency implement a subset of Biological Opinion projects that pertain to coho and the Water Agency is required to report progress on these efforts to CDFW.

Project implementation timelines in the Biological Opinion, and Consistency Determination, specify Water Agency reporting requirements to NMFS and CDFW and encourage frequent communication among the agencies. The Water Agency has engaged both NMFS and CDFW in frequent meetings and has presented project status updates on many occasions since early 2009. Although not an explicit requirement of the Biological Opinion or Consistency Determination, the Water Agency has elected to coalesce reporting requirements into one annual volume for presentation to the agencies. The following document represents the sixth report for year 2015-2016. Previous annual reports can be accessed at at the Water Agency's website: http://www.scwa.ca.gov.

Water Agency projects mandated by the Biological Opinion and Consistency Determination fall into six major categories:

- Biological and Habitat Monitoring;
- Habitat Enhancement;
- California Environmental Quality Act (CEQA) Compliance and Permitting;
- Planning and Adaptive Management;
- Water and Fish Facilities Improvements; and
- Public Outreach.

This report contains status updates for planning efforts, environmental compliance, and outreach but the majority of the technical information we present pertains to monitoring and habitat enhancement. The Biological Opinion requires extensive fisheries data collection in the mainstem Russian River, Dry Creek, and Estuary to detect trends and inform habitat enhancement efforts. The report presents each data collection effort independently and the primary intent of this document is to clearly communicate recent results. However, because Chinook salmon, coho salmon, and steelhead have complex life history patterns that integrate all of these environments, we also present a synthesis section to discuss the interrelated nature of the data. Some monitoring programs are extensions of ongoing Water Agency efforts that were initiated a decade or more before receipt of the Biological Opinion.

References

National Marine Fisheries Service (NMFS). 2008. Biological Opinion for Water Supply, Flood Control Operations, and Channel Maintenance conducted by the U.S. Army Corps of Engineers, the Sonoma County Water Agency, and the Mendocino County Russian River Flood Control and Water Conservation District in the Russian River Watershed. September 24, 2008.

CHAPTER 2 : Public Outreach

Biological Opinion Requirements

The Biological Opinion includes minimal *explicit* public outreach requirements. The breadth and depth of the RPAs, however, *implies* that implementation of the Biological Opinion will include a robust public outreach program.

RPA 1 (Pursue Changes to D1610 Flows) mandates two outreach activities. First, it requires the Water Agency, with the support of NMFS staff, to conduct outreach "to affected parties in the Russian River watershed" regarding permanently changing Decision 1610. Second, the RPA requires the Water Agency to update NMFS on the progress of temporary urgency changes to flows during Section 7 progress meetings and as public notices and documents are issued.

RPA 2 (Adaptive Management of the Outlet Channel) requires that within six months of the issuance of the Biological Opinion the Water Agency, in consultation with NMFS, "conduct public outreach and education on the need to reduce estuarine impacts by avoiding mechanical breaching to the greatest extent possible."

Finally, RPA 3 (Dry Creek Habitat Enhancements, refers to public outreach in the following mandate, "Working with local landowners, DFG[1] and NMFS, Water Agency will prioritize options for implementation" of habitat enhancement.

The remaining RPAs do not mention public outreach.

Water Agency Public Outreach Activities – 2015

Meetings

Public Policy Facilitating Committee (PPFC) meeting - The PPFC met in January 2015 for an update of the 2014 activities. Notices for the meeting were sent out to approximately 800 individuals and agencies and a press release was issued. Approximately 80 people attended the meeting and heard presentations from Josh Fuller, NMFS, Mike Dillabough, USACE and, from the Water Agency, Jessica Martini Lamb, Aaron Johnson, Gregg Horton, Dave Manning, Dave Cuneo, Steve Koldis, Ann DuBay, Justin Smith and Pam Jeane.

Community Meetings, Events & Tours – The seventh annual Russian River Estuary Lagoon Management Community Meeting was held in June 2015 at the Monte Rio Community Center. The meeting included discussions of this summer's Lagoon Management plans (Martini Lamb), results from 2014 water quality monitoring and plans for 2015 (Jeff Church), and a report of the jetty feasibility study (Dane Behrens & Matt Brennan, Environmental Science Associates-PWA). Immediately following the Estuary Meeting, a meeting was held regarding proposed Russian River flow levels. Jeane spoke about drought conditions, and the need to preserve water in

[1] DFG (Department of Fish and Game) is now known as the California Department of Fish and Wildlife.

Lake Mendocino. DuBay discussed outreach and water conservation efforts. About 80 people attended the meeting.

A community meeting on Dry Creek habitat enhancement was held in July 2015 at the Lake Sonoma Visitors Center. The meeting was co-hosted by the Dry Creek Valley Association, the Winegrape Growers of Dry Creek, the USACE and the Water Agency. Informational mailers were sent to more than 700 people and about 75 people attended the meeting to take a "virtual tour" of Dry Creek (Cuneo); hear about construction plans for summer 2015 (Greg Guensch); fish monitoring (Manning); conceptual plans for Miles 4 and 6 (Manning); and the Salmon Stewards program (Barry Dugan). Immediately following the Dry Creek meeting, a meeting was held regarding Russian River flow levels. Jeane spoke about drought conditions, and the need to preserve water in Lake Mendocino. Brad Sherwood discussed outreach and water conservation efforts.

Additional Dry Creek outreach included the Salmon Stewards of Dry Creek marketing program, the issuance of the first Dry Creek Bulletin, and the Dry Creek Habitat EIR.

The Salmon Steward program was promoted through materials and a hat for participants in habitat enhancement projects. The Fall 2015 Dry Creek Bulletin included articles about Phase 1 habitat enhancement projects, a profile of Don Wallace and Kim Stare Wallace and a description of the Salmon Stewards program.

The Dry Creek Habitat EIR outreach the community meeting, a press release and legal and display ads in regional and local newspapers (Press Democrat, Healdsburg Tribune and Windsor Times).

Tours held for public officials and others (coordinated with NMFS, DFG, Corps and Water Agency staff) included NOAA administrator Dr. Kathryn Sullivan, Will Stelle (NMFS Regional Administrator - West Coast Region (WCR), Irma Lagomarsino, (Assistant Regional Administrator - California Coastal Area Office, WCR), Dr. Rob Cifelli (Team lead, Hydrometeorology Forcing Science Team, PSD, Earth System Research Laboratory), Dr. Robin Webb, Dr. Roger Pulwarty, Alan Haynes (Service Coordination Hydrologist, California Nevada River Forecast Center). On separate tour, the following federal officials toured Dry Creek: Tom Champeau, (Vice Chair of the National Fish Habitat Board - Chief, Division of Freshwater Fisheries Management, Florida Fish and Wildlife Conservation Commission); Miranda Plumb, Acting Fish Passage and Habitat Partnerships Coordinator (U.S. Fish and Wildlife Service, Pacific Region); Andrei V. Rykoff, Timber Sale Prep and Stewardship Contracting Section Head, (Forest Service Pacific Southwest Region); and Dan Shively, National Fisheries Program Manager, Forest Service, Watershed, Fish, Wildlife, Air & Rare Plants).

The Water Advisory Committee and Technical Advisory Committee, attendees of the 2015 PPFC meeting, Congressman Jared Huffman and staff, and several small groups also toured habitat enhancement projects in 2015.

Other Outreach

Free Media – Several articles about Biological Opinion projects appeared in 2015 in The Press Democrat, the Russian River Times, the West County News and Review, and North Bay Bohemian, and the Russian River Gazette. In 2015, press releases were issued on Mirabel fishway construction, Dry Creek habitat construction, community meetings regarding the estuary and Dry Creek, Chinook returns, coho releases and the Public Policy Facilitating Committee meeting.

Electronic Media – The Water Agency continually updated its Biological Opinion webpage, including links on new documents and meetings. In addition, the Water Agency posted videos on YouTube regarding Dry Creek habitat construction, which can be accessed via the agency's website. Email alerts regarding activities in the estuary were issued about 10 times in 2015. Emails also were issued to neighbors regarding progress on the Mirabel Fish Passage Improvement Project.

Materials – In 2015, flyers regarding the Dry Creek Demonstration Project and the Mirabel Fish Passage Improvement Project were updated several times to reflect different stages of construction. Other materials were updated and distributed at meetings, conferences, statewide forums, outreach events and through the Water Agency website.

CHAPTER 3 : Pursue Changes to Decision 1610 Flows

Two major reservoir projects provide water supply storage in the Russian River watershed: 1) Coyote Valley Dam/Lake Mendocino, located on the East Fork of the Russian River three miles east of Ukiah, and 2) Warm Springs Dam/Lake Sonoma, located on Dry Creek 14 miles northwest of Healdsburg. The Water Agency is the local sponsor for these two federal water supply and flood control projects, collectively referred to as the Russian River Project. Under agreements with the USACE, the Water Agency manages the water supply storage space in these reservoirs to provide a water supply and maintain summertime Russian River and Dry Creek streamflows.

The Water Agency holds water-right permits[1] issued by the State Water Resources Control Board (SWRCB) that authorize the Water Agency to divert[2] Russian River and Dry Creek flows and to re-divert[3] water stored and released from Lake Mendocino and Lake Sonoma. The Water Agency releases water from storage in these lakes for delivery to municipalities, where the water is used primarily for residential, governmental, commercial, and industrial purposes. The primary points of diversion include the Water Agency's facilities at Wohler and Mirabel Park (near Forestville). The Water Agency also releases water to satisfy the needs of other water users and to contribute to the maintenance of minimum instream flow requirements in the Russian River and Dry Creek established in 1986 by the SWRCB's Decision 1610. These minimum instream flow requirements vary depending on specific hydrologic conditions (normal, dry, and critical) that are based on cumulative inflows into Lake Pillsbury in the Eel River watershed.

NMFS concluded in the Russian River Biological Opinion that the artificially elevated summertime minimum flows in the Russian River and Dry Creek currently required by Decision 1610 result in high water velocities that reduce the quality and quantity of rearing habitat for coho salmon and steelhead. NMFS' Russian River Biological Opinion concludes that reducing Decision 1610 minimum instream flow requirements will enable alternative flow management scenarios that will increase available rearing habitat in Dry Creek and the upper Russian River, and provide a lower, closer-to-natural inflow to the estuary between late spring and early fall, thereby enhancing the potential for maintaining a seasonal freshwater lagoon that would likely support increased production of juvenile steelhead and salmon.

Changes to Decision 1610 are under the purview of the SWRCB, which retained under Decision 1610 the jurisdiction to modify minimum instream flow requirements if future fisheries studies identified a benefit. NMFS recognized that changing Decision 1610 would require a multi-year (6

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[1] SWRCB water-right permits 12947A, 12949, 12950 and 16596.
[2] Divert – refers to water diverted directly from streamflows into distribution systems for beneficial uses or into storage in reservoirs.
[3] Re-divert – refers to water that has been diverted to storage in a reservoir, then is released and diverted again at a point downstream.

to 8 years) process of petitioning the SWRCB for changes to minimum instream flow requirements, public notice of the petition, compliance with CEQA, and a SWRCB hearing process. To minimize the effects of existing minimum instream flows on listed salmonids during this process, the Russian River Biological Opinion stipulated that the Water Agency "will seek both long term and interim changes to minimum flow requirements stipulated by D1610." The permanent and temporary changes to Decision 1610 minimum instream flow requirements specified by NMFS in the Russian River Biological Opinion are summarized in Figure 3.1.

Permanent Changes

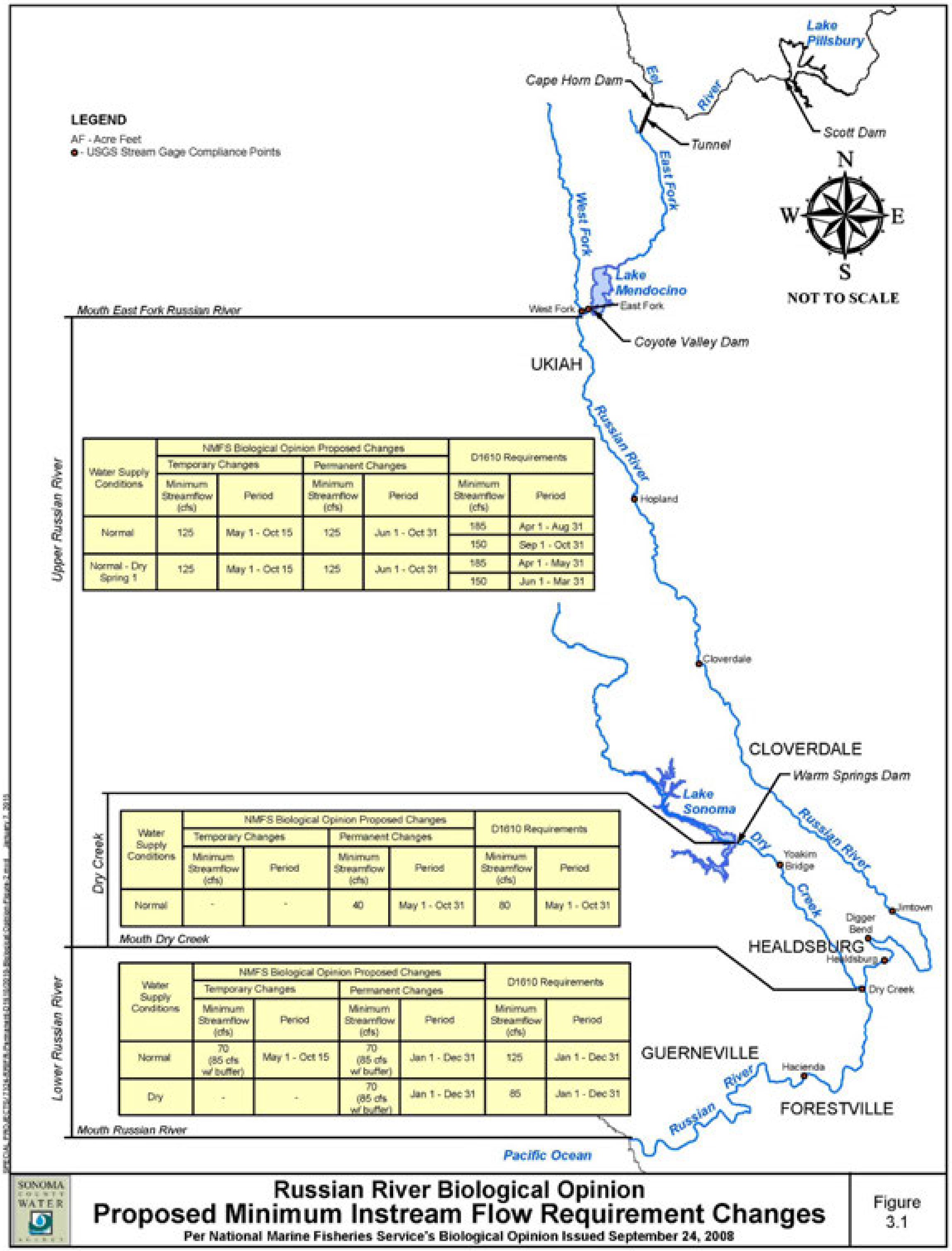
The Russian River Biological Opinion requires the Water Agency to begin the process of changing minimum instream flows by submitting a petition to change Decision 1610 to the SWRCB within one year of the date of issuance of the final Biological Opinion. The Water Agency filed a petition with the SWRCB on September 23, 2009, to permanently change Decision 1610 minimum instream flow requirements. The requested changes are to reduce minimum instream flow requirements in the mainstem Russian River and Dry Creek between late spring and early fall during normal and dry water years and promote the goals of enhancing salmonid rearing habitat in the upper Russian River mainstem, lower river in the vicinity of the Estuary, and Dry Creek downstream of Warm Springs Dam. NMFS' Russian River Biological Opinion concluded that, in addition to providing fishery benefits, the lower instream flow requirements "should promote water conservation and limit effects on in-stream river recreation." NMFS stated that the following changes, based on observations during the 2001 interagency flow-habitat study and the 2007 low flow season, may achieve these goals:

During Normal Years:

1. Reduce the minimum flow requirement for the Russian River from the East Fork to Dry Creek from 185 cubic-feet per second (cfs) to 125 cfs between June 1 and August 31; and from 150 cfs to 125 cfs between September 1 and October 31.
2. Reduce the minimum flow requirement for the Russian River between the mouth of Dry Creek and the mouth of the Russian River from 125 cfs to 70 cfs.
3. Reduce the minimum flow requirement for Dry Creek from Warm Springs Dam to the Russian River from 80 cfs to 40 cfs from May 1 to October 31.

During Dry Years:

1. Reduce the minimum flow requirement for the Russian River between the mouth of Dry Creek and the mouth of the Russian River from 85 cfs to 70 cfs.



Upper Russian River

Water Supply Conditions	NMFS Biological Opinion Proposed Changes				D1610 Requirements	
	Temporary Changes		Permanent Changes			
	Minimum Streamflow (cfs)	Period	Minimum Streamflow (cfs)	Period	Minimum Streamflow (cfs)	Period
Normal	125	May 1 - Oct 15	125	Jun 1 - Oct 31	185	Apr 1 - Aug 31
					150	Sep 1 - Oct 31
Normal - Dry Spring 1	125	May 1 - Oct 15	125	Jun 1 - Oct 31	185	Apr 1 - May 31
					150	Jun 1 - Mar 31

Dry Creek

Water Supply Conditions	NMFS Biological Opinion Proposed Changes				D1610 Requirements	
	Temporary Changes		Permanent Changes			
	Minimum Streamflow (cfs)	Period	Minimum Streamflow (cfs)	Period	Minimum Streamflow (cfs)	Period
Normal	-	-	40	May 1 - Oct 31	80	May 1 - Oct 31

Lower Russian River

Water Supply Conditions	NMFS Biological Opinion Proposed Changes				D1610 Requirements	
	Temporary Changes		Permanent Changes			
	Minimum Streamflow (cfs)	Period	Minimum Streamflow (cfs)	Period	Minimum Streamflow (cfs)	Period
Normal	70 (85 cfs w/ buffer)	May 1 - Oct 15	70 (85 cfs w/ buffer)	Jan 1 - Dec 31	125	Jan 1 - Dec 31
Dry	-	-	70 (85 cfs w/ buffer)	Jan 1 - Dec 31	85	Jan 1 - Dec 31

Figure 3.1. A summary of the permanent and temporary changes to Decision 1610 minimum instream flow requirements specified by NMFS in the Russian River Biological Opinion.

Summary Status

The SWRCB issued a second amended public notice of the Water Agency's petition to modify Decision 1610 for public comment on March 29, 2010. Following filing of the petition to change Decision 1610, the Water Agency issued a Notice of Preparation (NOP) of an Environmental Impact Report (EIR) for the Fish Habitat Flows and Water Rights Project (Fish Flow Project).

Temporary Changes

Until the SWRCB issues an order on the petition to permanently modify Decision 1610, the minimum instream flow requirements specified in Decision 1610 (with the resulting adverse impacts to listed salmonids) will remain in effect, unless temporary changes to these requirements are made by the SWRCB. The Russian River Biological Opinion requires that the Water Agency petition the SWRCB for temporary changes to the Decision 1610 minimum instream flow requirements beginning in 2010 and for each year until the SWRCB issues an order on the Water Agency's petition for the permanent changes to these requirements. NMFS' Russian River Biological Opinion only requires that petitions for temporary changes "request that minimum bypass flows of 70 cfs be implemented at the USGS gage at the Hacienda Bridge between May 1 and October 15, with the understanding that for compliance purposes SCWA will typically maintain about 85 cfs at the Hacienda gage. For purposes of enhancing steelhead rearing habitats between the East Branch and Hopland, these petitions will request a minimum bypass flow of 125 cfs at the Healdsburg gage between May 1 and October 15."

Summary Status

The Water Agency submitted a Temporary Urgency Change Petition to the SWRCB on April 21, 2015, to preserve the drought-limited water supply in Lake Mendocino (Appendix 3.1). The SWRCB issued an Order approving the Water Agency's TUCP on May 1, 2015 (Appendix 3.2). The Water Agency submitted a request to amend the Order on May 27, 2015 (Appendix 3.3). The Order was modified on June 17, 2015, due to the ongoing drought conditions and in accordance with the Governor's Drought State of Emergency declaration (Appendix 3.4).

The SWRCB's modified Order made the following changes to the Water Agency's permits until October 27, 2015: minimum instream flow in the upper Russian River (from its confluence with the East Fork of the Russian River to its confluence with Dry Creek) remained at or above 75 cfs through June 15, 2015 and remained at or above 25 cfs starting June 16, 2015; and minimum instream flow in the lower Russian River (from its confluence with Dry Creek to the Pacific Ocean) remained at or above 85 cfs through June 15, 2015 and remain at or above 50 cfs starting June 16, 2015. To allow the Water Agency to optimally manage flows in the Upper Russian River and Lower Russian River, the modified Order allowed for the use of 24-hour mean instream flow criterion.

The modified Order included several terms and conditions, including requirements for fisheries habitat monitoring and regular consultation with National Marine Fisheries Service and California Department of Fish and Wildlife regarding fisheries conditions (Terms 2 to 7), preparation of a water quality monitoring plan and summary data report (Terms 10 to 14), reporting on hydrologic conditions of the Russian River system (Term 15), reporting of activities and programs implemented by the Water Agency and its contractors to assess and reduce water loss and promote increasing water use efficiency (Term 18), and operations in accordance with a Water Demand Reduction Plan (Term 20).

Reports to fulfill the terms of the Order were prepared and submitted to the SWRCB and are provided in Appendix 3.5. The reports included: Provision 7 – Fisheries Monitoring Tasks; Term 20 –Implementation of Conservation Regulatory Framework (for Order dated May 1, 2015); Term 20 – Implementation of Conservation Regulatory Framework (dated June 17, 2015); and Provision 17 -Water Demand Reduction Plan.

Provisions 2 through 7 of the State Water Board Order required the Water Agency to conduct and report on fisheries conditions. Updates of fisheries monitoring and consultation status were sent to NMFS and CDFW staff every two weeks per the State Water Board Order.

The Water Agency conducted weekly bacteriological, nutrient and algal mainstem sampling at five sites in the Russian River Estuary. All samples were analyzed for nutrients, chlorophyll *a*, standard bacterial indicators (total coliforms, *E. coli* and enterococci), total and dissolved organic carbon, turbidity, and total dissolved solids. Bacteria analysis for the Water Agency was conducted by the Sonoma County DHS Public Health Division Lab in Santa Rosa. *E. coli* and total coliform were analyzed using the Colilert method and enterococcus was analyzed using the Enterolert method. In addition, data sondes monitoring temperature, dissolved oxygen, pH, and specific conductance were operated at multiple stations from Ukiah to Jenner.

Monitoring results were posted to the Water Agency website and are provided in Appendix 3.6. Water quality monitoring in the Russian River Estuary is further discussed in Chapter 4.

CHAPTER 4 Estuary Management

The Russian River estuary (Estuary) is located approximately 97 kilometers (km; 60 miles) northwest of San Francisco in Jenner, Sonoma County, California. The Estuary extends from the mouth of the Russian River upstream approximately 10 to 11 km (6 to 7 miles) between Austin Creek and the community of Duncans Mills (Heckel 1994). When a barrier beach forms and closes the river mouth, a lagoon forms behind the beach and reaches up to Vacation Beach.

The Estuary may close throughout the year as a result of a barrier beach forming across the mouth of the Russian River. The mouth is located at Goat Rock State Beach (California Department of Parks and Recreation). Although closures may occur at any time of the year, the mouth usually closes during the spring, summer, and fall (Heckel 1994; Merritt Smith Consulting 1997, 1998, 1999, 2000; Sonoma County Water Agency and Merritt Smith Consulting 2001). Closures result in ponding of the Russian River behind the barrier beach and, as water surface levels rise in the Estuary, flooding may occur. The barrier beach has been artificially breached for decades; first by local citizens, then the County of Sonoma Public Works Department, and, since 1995, by the Water Agency. The Water Agency’s artificial breaching activities are conducted in accordance with the Russian River Estuary Management Plan recommended in the Heckel (1994) study. The purpose of artificially breaching the barrier beach is to alleviate potential flooding of low-lying properties along the Estuary.

The National Marine Fisheries Service’s (NMFS) Russian River Biological Opinion (NMFS 2008) found that artificially elevated inflows to the Russian River estuary during the low flow season (May through October) and historic artificial breaching practices have significant adverse effects on the Russian River’s estuarine rearing habitat for steelhead, coho salmon, and Chinook salmon. The historical method of artificial sandbar breaching, which is done in response to rising water levels behind the barrier beach, adversely affects the Estuary’s water quality and freshwater depths. The historical artificial breaching practices create a tidal marine environment with shallow depths and high salinity. Salinity stratification contributes to low dissolved oxygen at the bottom in some areas. The Biological Opinion (NMFS 2008) concludes that the combination of high inflows and breaching practices impact rearing habitat because they interfere with natural processes that cause a freshwater lagoon to form behind the barrier beach. Fresh or brackish water lagoons at the mouths of many streams in central and southern California often provide depths and water quality that are highly favorable to the survival of rearing salmon and steelhead.

The Biological Opinion’s RPA 2, Alterations to Estuary Management, (NMFS 2008) requires the Water Agency to collaborate with NMFS and to modify Estuary water level management in order to reduce marine influence (high salinity and tidal inflow) and promote a higher water surface elevation in the Estuary (formation of a fresh or brackish lagoon) for purposes of enhancing the quality of rearing habitat for young-of-year and age 1+ juvenile (age 0+ and 1+) steelhead from May 15 to October 15 (referred to hereafter as the “lagoon management period”). A program of potential, incremental steps are prescribed to accomplish this, including adaptive management of a lagoon outlet channel on the barrier beach, study of the existing jetty and its potential influence on beach formation processes and salinity seepage through the barrier beach, and a feasibility study of alternative flood risk measures. RPA 2 also includes provisions for monitoring the response of water quality, invertebrate production, and salmonids in the Estuary to the management of water surface elevations during the lagoon management period.

The following section provides a summary of the Water Agency's estuary management actions required under the Russian River Biological Opinion RPA 2 in 2015. These actions are also required by other regulatory permits issued for the Estuary Management Project, including the California Coastal Commission's Coastal Development Permit (CDP) and North Coast Regional Water Quality Control Board Clean Water Act Section 401 Water Quality Certification (Certification). References to the Biological Opinion's RPA are used to maintain consistency with previous annual reports.

One of the conditions in the Coastal Commission CDP is to prepare a Water Quality Monitoring Plan (Monitoring Plan) for the Russian River Estuary. The objectives of the Monitoring Plan are to provide information to evaluate potential changes to water quality and availability of habitat for aquatic resources resulting from the proposed changes to management of the Estuary as a seasonal freshwater lagoon from May 15 to October 15 (lagoon management period) with a low-velocity outlet channel as required by the Biological Opinion. Furthermore, the Monitoring Plan will build upon previous water quality studies that have been conducted in the Estuary as required by the Russian River Biological Opinion, TUC Petitions, and the Stipulated Judgment.

In addition, the NCRWQCB issued Clean Water Act (CWA) section 401 water quality certification (Certification) permit number WDID 1B10122WNSO for the Estuary Project on May 14, 2014. The conditions of the permit require a monitoring and reporting plan as well as additional focused water quality sampling related to contact recreation in the Russian River Estuary and maximum backwater area between Jenner and Vacation Beach.

Regarding water quality monitoring to support the Russian River Biological Opinion, TUC, Stipulated Judgment, CDP, and Water Quality Certification for Estuary management, the following questions help to explain the objective of the monitoring plan:

- What are the background levels of nutrients and pathogens in the Estuary under open, tidally influenced conditions? How do these background levels respond to changes in managing the Estuary as a seasonal freshwater lagoon, considering other contributing factors?
- Do water temperature, dissolved oxygen, and salinity respond to changes managing the Estuary as a seasonal freshwater lagoon?
- Are there secondary biological effects related to changes in water quality from managing the Estuary as a seasonal freshwater lagoon (e.g. stress to fish, plants, invertebrates) and if so, what are they?
- Are there affects to public health/recreation?

Barrier Beach Management

RPA 2 requires the Water Agency, in coordination with NMFS, California Department of Fish and Wildlife (CDFW), and the U.S. Army Corps of Engineers (USACE), to annually prepare barrier beach outlet channel design plans. Each year after coordinating with the agencies, the Water Agency is to provide a draft plan to NMFS, CDFW, and the USACE by April 1 for their review and input. The initial plan was to entail the design of a lagoon outlet channel cut diagonally to the northwest. Sediment transport equations shall be used by Water Agency as channel design criteria to minimize channel scour at the anticipated rate of Russian River discharge. This general channel design will be used instead of traditional mechanical breaching whenever the barrier beach closes and it is safe for personnel and equipment to work on the barrier beach. Alternate methods may include 1) use of a channel cut to the south if prolonged south west swells occur, and 2) use of the current jetty as a channel grade control structure (as described below) for maintaining water surface elevations up to 7-9 feet NGVD (NMFS 2008).

The Water Agency contracted with Environmental Science Associates (ESA PWA) to prepare the Russian River Estuary Outlet Channel Adaptive Management Plan (Appendix 4.1). The approach of the plan was to meet the objective of RPA 2 to the greatest extent feasible while staying within the constraints of existing regulatory permits and minimizing the impact to aesthetic, biological, and recreational resources of the site. It was recognized that the measures developed in the management plan, when implemented, potentially could not fully meet the objectives established by the RPA. The concept of this approach was developed in coordination with NMFS, CDFW, and California State Parks (State Parks). The annual meeting with regulatory agency staff to discuss the prior year's beach management activities and preparation of the updated 2015 annual Outlet Channel Adaptive Management Plan was held on April 9, 2015. In attendance were staff from the Water Agency, ESA PWA, University of California, Davis's Bodega Marine Laboratory (Bodega Marine Lab), NMFS, CDFW, North Coast Regional Water Quality Control Board (NCRWQCB), and the Lawrence Berkeley National Laboratory. Only minor updates to the prior year's plan were made in the 2015 plan, which includes a summary of physical processes during 2011, 2012, 2013, and 2014 as Appendices F, G, H, and I, respectively. Only minor updates to the prior year's plan were made in the 2015 plan, which includes a summary of physical processes during 2011, 2012, 2013, and 2014 as Appendices F, G, H, and I, respectively. The revised plan was in effect for 2015, but no opportunities for management action occurred during the management period. Outlet channel implementation has occurred only in 2010 and is summarized in Appendix F of the 2015 Outlet Channel Adaptive Management Plan (Appendix 4.1).

A monthly topographic survey of the beach at the mouth of the Russian River is also required under RPA 2. Topographic data was collected monthly in 2015 and provided to NMFS and CDFW. The April 2015 topographic survey was scheduled twice that month, but canceled due to the presence of neonate harbor seals at the mouth of the Russian River. The December 2015 topographic survey was not performed due to hazardous beach conditions and storm events that month. The beach topographic maps are provided in Appendix 4.2.

ESA prepared the 2016 Russian River Estuary Outlet Channel Adaptive Management Plan (Appendix 4.3). The approach of the plan was to meet the objective of RPA 2 as described previously. The annual meeting with regulatory agency staff to discuss the prior year's beach management activities and preparation of the updated 2016 annual Outlet Channel Adaptive Management Plan was held on March 14, 2016. In attendance in person and by conference line were staff from the Water Agency, ESA PWA, Bodega Marine Lab, NMFS, CDFW, NCRWQCB, State Parks, California State Lands Commission, and Lawrence Berkeley National Laboratory.

As described in Appendix K of the 2016 Outlet Channel Adaptive Management Plan, during the 2015 management period, May 15th to October 15th, Water Agency staff regularly monitored current and forecasted Estuary water levels, inlet state, river discharge, tides, and wave conditions to anticipate changes to the inlet's state. Although a 20-day closure event began in late May, the mouth self-breach before an outlet channel could be created. The estuary was then tidal for several month until it closed again in early September for the first of two, approximately month-long closures. The closure starting on September 8 self-breached on October 3 before water surface elevations reached 7 feet at the Jenner gage. The closure starting on October 10 continued until November 5, outside of the management period, and ended with an artificial breaching (ESA PWA 2016).

Lagoon Management Season Closures and Self-Breaches

Time series of Estuary water levels, as well as the key forcing factors (waves, tides, and riverine discharge), are shown in Figure 4.1 for the entire management period (ESA PWA 2016). The lagoon water level time series (Figure 4.1a) summarizes the closure events at the beginning of the management period, as well as the subsequent tidal conditions and later closure events in fall (Figure 4.2). As shown in Figure 4.1d, flows at Guerneville dropped to 100 ft3/s by roughly July 1st, which was more than a month later than in 2014. These higher flows contributed to the rate of water surface elevation increase during the May-June closure event. During this closure, construction equipment access could not access the beach north of the groin due to the lagoon's position and the steep drop-off on the north side of the groin (Figure 4.3). Therefore, no beach management was scheduled and the lagoon filled to the beach crest and self-breached. From July to October, flows were mostly below 100 ft3/s, and dipped below 70 ft3/s for parts of late July, September and October. As in prior years, wave energy was minimal through the summer months. Since waves were derived in 2015 from the Point Arena buoy instead of the Point Reyes buoy, and both of these buoys were off-line after mid-September, only a qualitative assessment of the events causing closure in 2015 was made. In prior years, closure events typically coincided with either moderately high waves (Hs > 6 ft) having periods greater than 10 s, or with neap oceanic tide ranges of less than approximately 5 ft. The May-June closure event happened during a neap tide cycle but during a period of relatively weak (Hs < 5 ft), but long period (~15 sec) waves. Moderately high waves and a neap tide cycle coincided with the closure event that began on September 8.

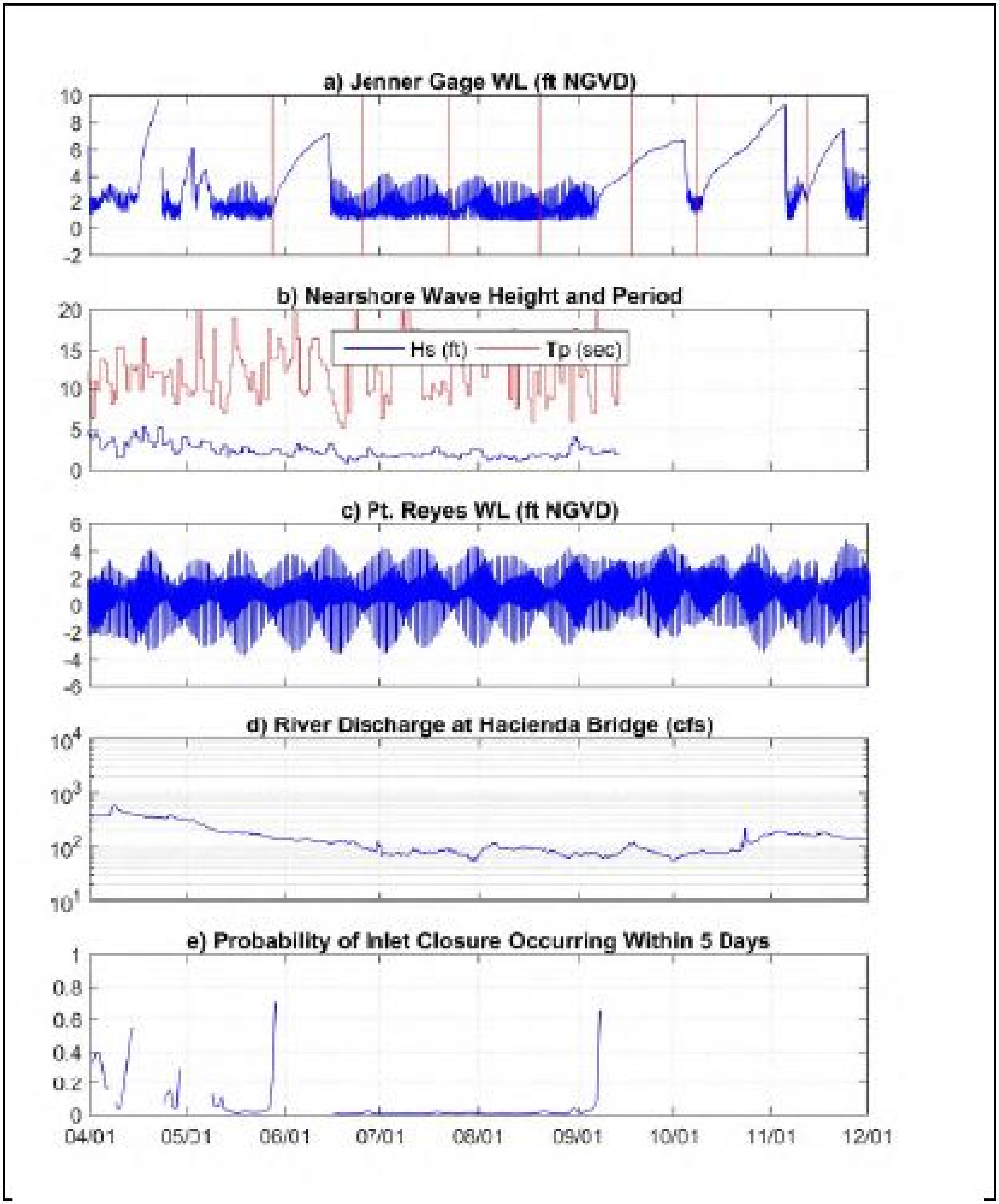


Figure 4.1. Estuary, Ocean, and River Conditions Compared with Closure Probability: April – November 2015.

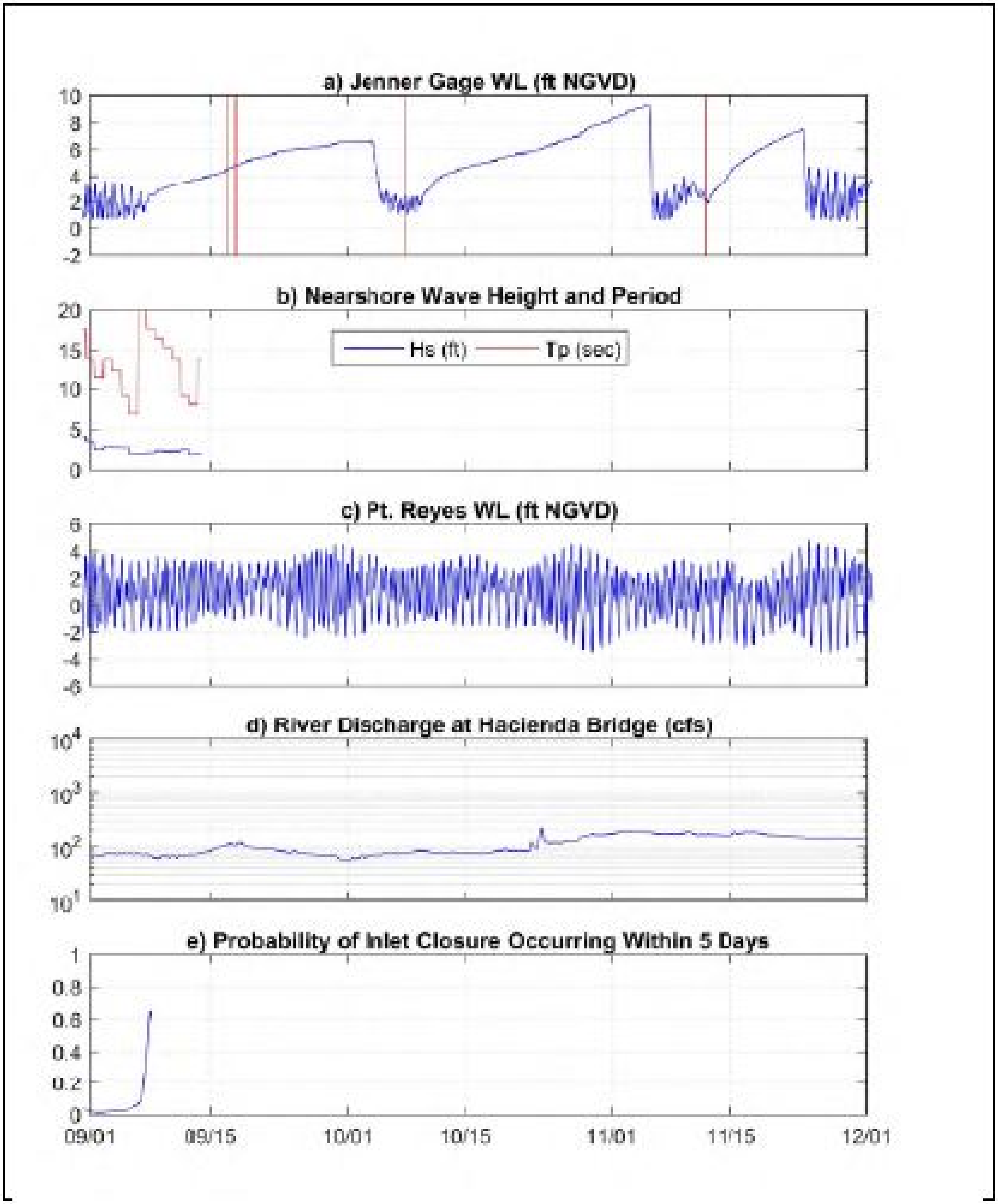


Figure 4.2. Estuary, Ocean, and River Conditions Compared with Closure Probability: September – November 2015.

Figure 4.3. Blocked beach access during closures a) June 4, 2015; b) September 29, 2015.

Appendix K of the 2016 Russian River Estuary Outlet Channel Adaptive Management Plan offers lessons learned based on 2015 observations of the Estuary, associated physical processes, and the Water Agency's planning for outlet channel management. These are summarized here and may be found in Appendix 4.3 of this report for fuller context:

- The beach north of the inlet remained steady between 11 and 15 ft NGVD. This was lower than previous years since the inlet migrated north in early winter and later migrated south to the groin. Near the groin, the berm was lowered by inlet migration when not undergoing beach building.
- The inlet returned to the groin in late winter, much earlier than in most years. This inlet alignment is not common, but has been observed in past years (Behrens et al., 2009).
- Peak annual river discharge has remained below 43,000 ft3/s for 9 consecutive years, a streak unmatched in the 70-year flow record. This lack of larger fluvial discharge may contribute to the predominant inlet location near the groin.
- The beach width in 2015 at Transect 3 (near Haystack Rock) was larger than in 2014. This may suggest that beach width is closely tied to inlet migration – the lack of migration north of Haystack Rock for several years has allowed the beach to grow at this end of the littoral cell.

Artificial Breaching

Outside of the management season, there were seven mouth closures in 2015. The Water Agency artificially breached (breaching) the barrier beach at the Russian River mouth outside the lagoon management period three times in 2015. The breachings were necessary to minimize flood risk to low-lying structures, which occurs at or above an elevation of approximately 9 feet NGVD at the Jenner gage located at State Parks' Jenner visitor center. No beach management activities occurred during the lagoon management period (May 15 – October 15).

The methods to artificially breach the barrier beach followed all state and federal permit requirements. These requirements included notification to State Parks' District headquarters, Sonoma Coast lifeguards, Monte Rio Fire Department, postings at Goat Rock State Beach and the State Parks' visitors center in Jenner (the Water Agency also placed public notifications at seven additional locations in the Estuary area); restricting equipment and activities to the breaching area; removal of equipment daily; and pinniped monitoring before, during, and after breaching.

Dune habitat and pinniped monitoring followed permit requirements from the California State Lands Commission, California Coastal Commission, CDFW, State Parks, NCRWQCB, USACE, and NMFS. No vegetation was disturbed and no animals were injured or killed. Pinniped monitoring followed procedures required by the Marine Mammal Protection Act Incidental Harassment Authorization issued by the NMFS for the Estuary Management Project.

The river mouth closed on March 27, 2015, and was breached on March 31. There were two attempts at breaching in November. The first breach on November 2 ended in a closure and the barrier beach was successfully breached on November 5. The river mouth closed again on

November 13 and was artificially breached on November 23. A closure event that began on December 2 led to flooding in Jenner. After the mouth closed, wave overwash and river discharge rapidly increased the water levels in the lagoon. Wave overwash conditions made the beach inaccessible to construction equipment for several days starting on December 8, preventing safe access to the beach for artificial breaching. Water surface elevations reached an estimated peak of 12.25 feet NGVD29 before the Estuary self-breached on December 13.

The Water Agency conducted three breaching attempts during spring and fall 2015 (Table 1; Figure 4.4). Time series photographs of each breaching event are shown in Figures 4.5 – 4.8. One mouth closure occurred in March. During fall closures in October and November 2015.

A pre-construction field meeting to discuss pinniped haulouts, permit conditions, and safety issues was held at the Highway 1 overlook in the morning with Water Agency staff prior to staff entering the beach (Figure 4.4) for each breaching event. Project activities were monitored by the project manager, breaching crew lead staff, and biological monitor at the Highway 1 overlook and were in radio contact with the breaching crew on the beach.

The Water Agency breaching crew was comprised of the equipment operator, two staff on foot monitoring safety conditions, and an additional staff member near the jetty and work area boundary to talk with any beach visitors. The excavator was escorted from the Goat Rock State Beach parking lot across the unvegetated sandbar to the river mouth. Excavation of a pilot channel across the sandbar took about 1 to 4 hours to complete, depending on the size of the barrier beach and water surface elevations. The excavator and field crew departed the beach once the barrier beach was breached.

Staff and equipment cautiously and slowly approached the breaching site and harbor seal haulout. The locations of harbor seal haulouts and numbers of seals are shown on Figures 4.5 through 4.8. Following a breaching event harbor seals returned to a haulout (usually at the location of the constructed pilot channel) within a day after a breach. Harbor seal numbers the day after breaching were similar, or higher, than observed prior to breaching. No seal pups were observed on the beach during any breaching event.

Table 4.1: Summary of beach management activities at Goat Rock State Beach for the Russian River Estuary Management Project, 2015. Location of activities are shown on Figure 4.4.

Closure Date	Beach Management Date	No. Days Closed	Activity Time[1]	Water Elevation (ft)[2]	Beach Management Activity[3]	Excavated Volume (CY)[4]
27-Mar	31-Mar	4	10:19am-10:39am	8.80	Pilot Channel	132
10-Oct	2-Nov	23	11:10am-12:54pm	8.68	Pilot Channel	775
2-Nov	5-Nov	3	9:12am-10:26am	9.31	Pilot Channel	495
13-Nov	23-Nov	10	9:21am-1:03pm	7.46	Pilot Channel	1,220

[1] Estimated period that excavator/bulldozer equipment was on the beach.
[2] Water surface elevation recorded at the Jenner gage located at the Jenner Visitor's Center.
[3] Beach management activity consists of a pilot channel to initiate an artificial breach of the barrier beach or outlet channel to form a lagoon.
[4] Estimated volume of sand excavated with heavy equipment during artificial breach or lagoon management activity.

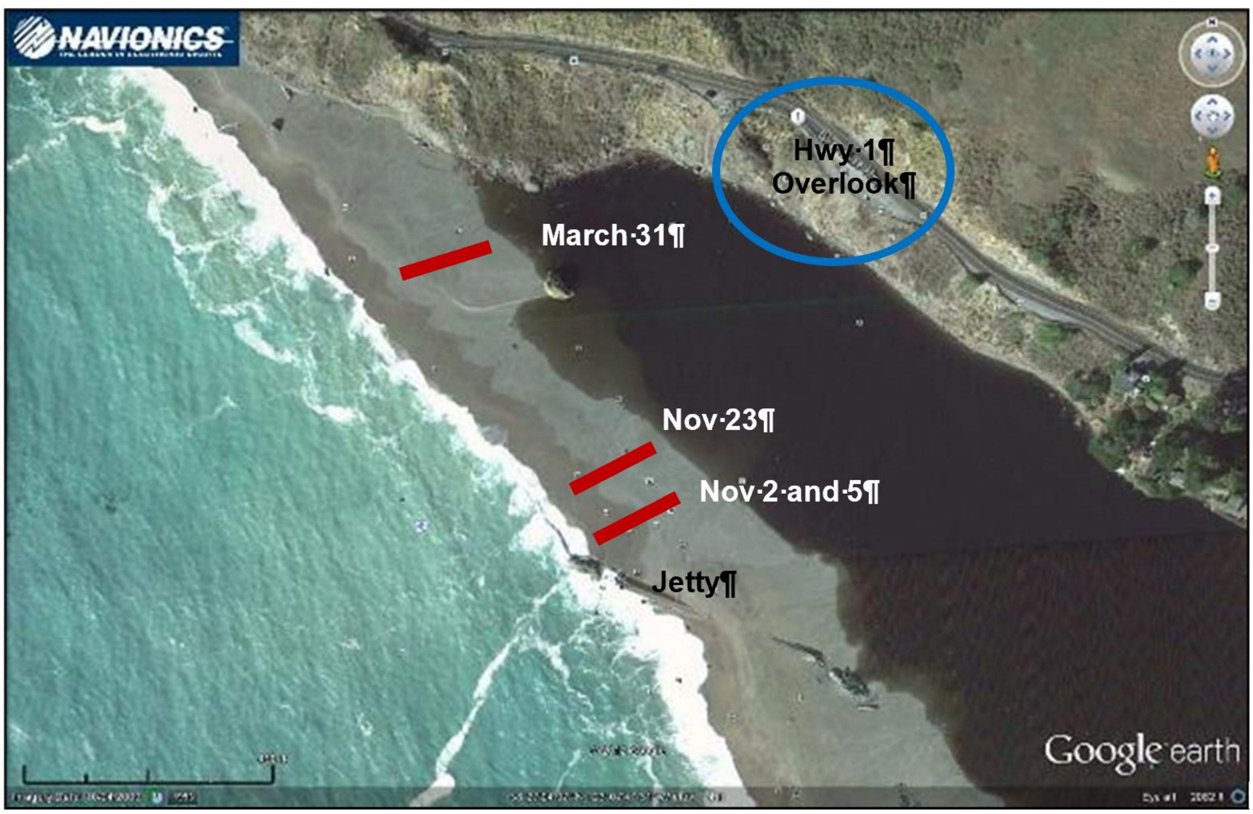


Figure 4.4. Russian River at the Pacific Ocean, Goat Rock State Beach. General location of artificial breaching pilot channel excavations in 2015.

Figure 4.5. Artificial breaching at the mouth of the Russian River Estuary, March 31, 2015. Photographs show pre- through post-breaching conditions. Excavation of a pilot channel took 20 minutes and was not captured on the timed photograph series.

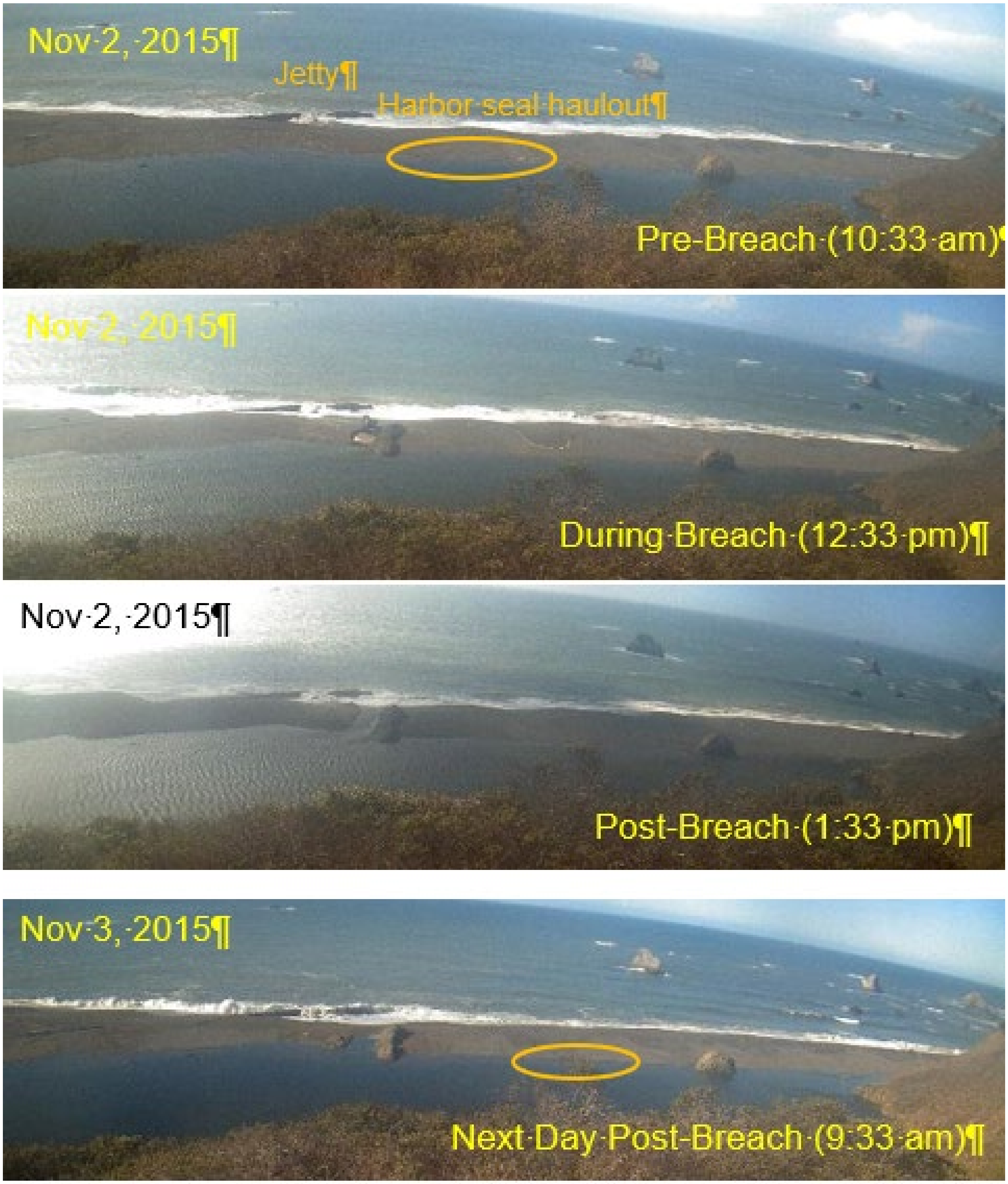


Figure 4.6. Artificial breaching at the mouth of the Russian River Estuary, November 2, 2015. Photographs show pre- through post- breaching conditions. The pilot channel closed soon after it was excavated.

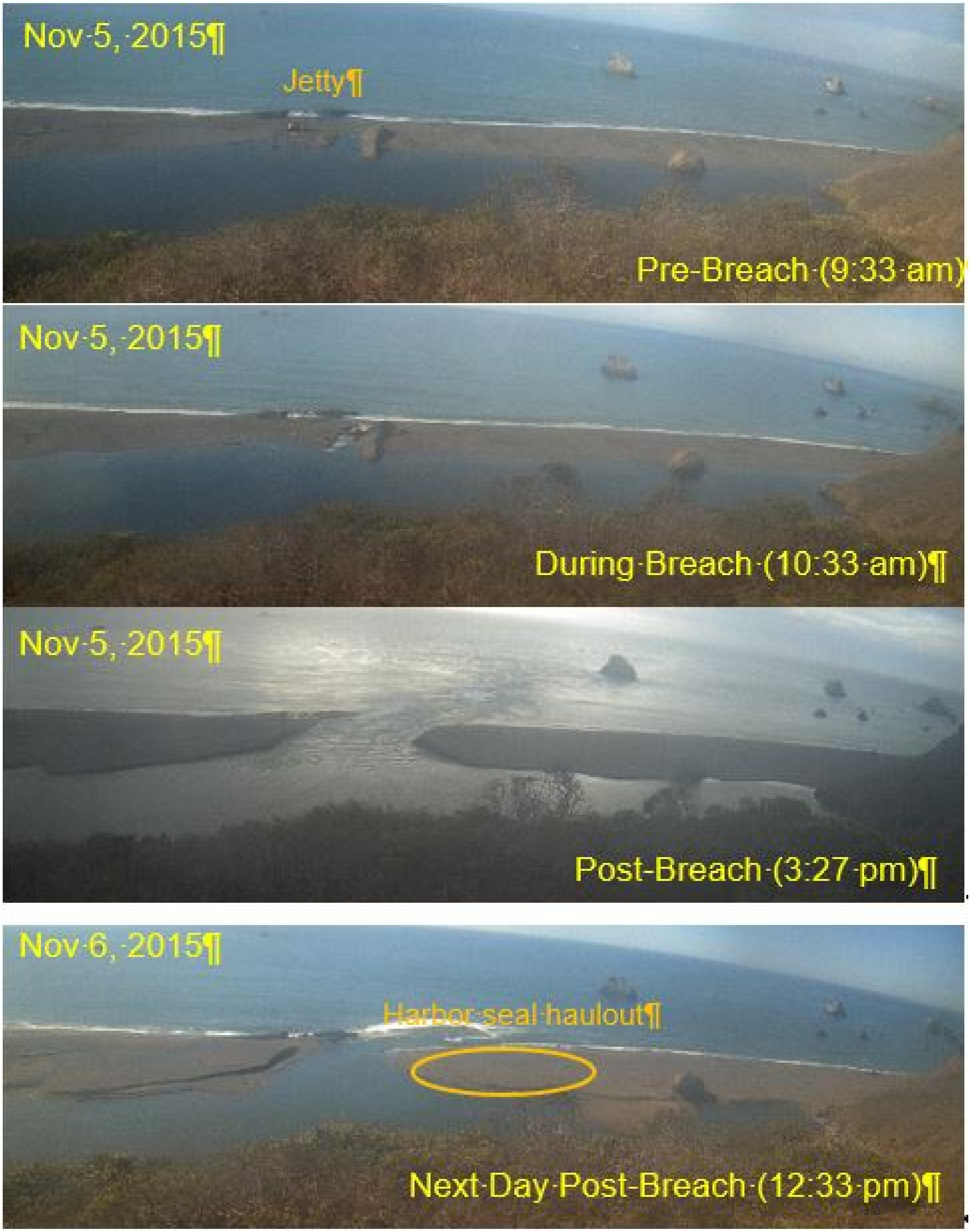


Figure 4.7. Artificial breaching at the mouth of the Russian River Estuary, November 5, 2015. Photographs show pre- through post-breaching conditions. Top photograph shows side cast sand from previous breaching attempt on November 2.

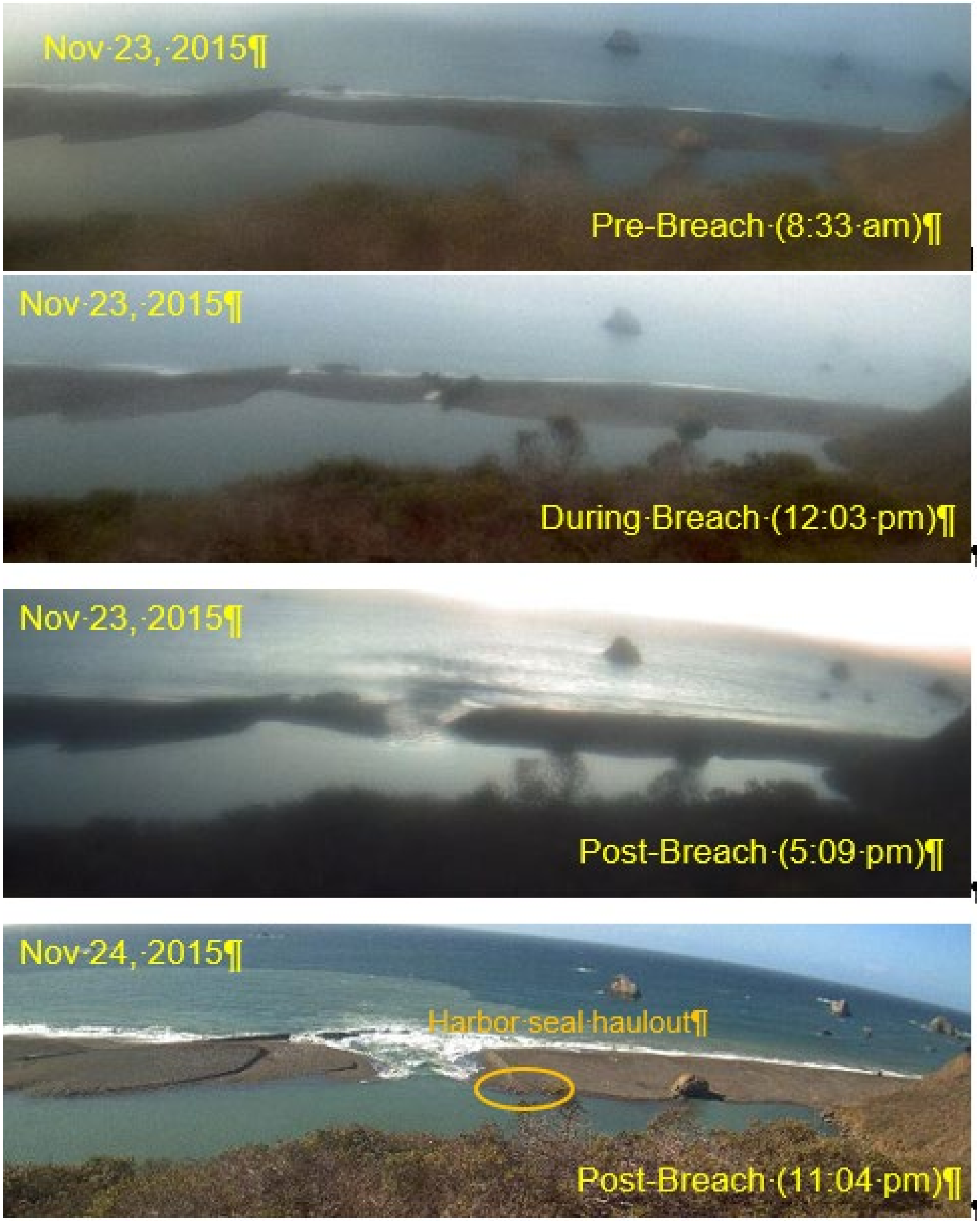


Figure 4.8. Artificial breaching at the mouth of the Russian River Estuary, November 23, 2015. Photographs show pre- through post-breaching conditions.

Pinniped Annual Monitoring

An Incidental Harassment Authorization (IHA) was issued by the NMFS pursuant to Section 101(a)(5)(D) of the Marine Mammal Protection Act (16 U.S.C 1361 et seq.) to take small numbers of marine mammals, by Level B harassment, incidental to the Water Agency's Estuary Management Project (issued April 20, 2015, original authorization dated March 30, 2010, NMFS IHA). An annual report of results of monitoring activities was submitted to NMFS and is provided in Appendix 4.4. A summary of the results of 2015 pinniped monitoring as reported in the *Russian River Estuary Management Project, Marine Mammal Protection Act Incidental Harassment Authorization, Report of Activities and Monitoring Results – January 1 to December 31, 2015* (SCWA 2016) are provided below.

Harbor seals (*Phoca vitulina richardsi*) regularly haul out at the mouth of the Russian River (Jenner haul-out). California sea lions (*Zalophus californianus*) and northern elephant seals (*Mirounga angustirostris*) are occasionally observed at the haul-out. There are also several known resting areas in the river at logs and rock piles. The Water Agency applied for an IHA under the MMPA for activities associated with Estuary management activities, which occur in the vicinity of these haul-outs, including:

- excavation and maintenance of a lagoon outlet channel that would facilitate management of a summer lagoon to improve rearing habitat for listed steelhead as required by the Russian River Biological Opinion (NMFS 2008);
- artificially breaching the barrier beach to minimize the potential for flooding of low-lying properties along the Estuary;
- biological and geophysical monitoring activities associated with the management actions described above;
- construction and maintenance of monitoring wells on the barrier beach south of the jetty; and
- geophysical surveys conducted at the barrier beach.

Pinniped monitoring was performed in accordance with the requirements of the NMFS IHA issued April 20, 2015, and the Russian River Estuary Management Activities Pinniped Monitoring Plan (Sonoma County Water Agency and Stewards of the Coast and Redwoods 2011).

In an attempt to understand possible relationships between use of the Jenner haul-out and nearby coastal and river (peripheral) haul-outs, several other haul-outs on the coast and in the Estuary were monitored. These haul-outs include North Jenner and Odin Cove to the north, Pocked Rock, Kabemali, and Rock Point to the south, and Penny Logs, Paddy's Rock, and Chalanchawi in the Estuary.

Baseline monitoring was performed to gather additional information about the population of harbor seals utilizing the Jenner haul-out including population trends, patterns in seasonal abundance and the influence of barrier beach condition on harbor seal abundance. Pinniped monitoring was also conducted in relation to Water Agency water level management events (lagoon outlet channel implementation and artificial breaching). Each of the peripheral haul-outs was monitored concurrent with Jenner baseline monitoring and monitoring of water level management activities. Estuary management monitoring occurred during the Water Agency's monthly topographic surveys of the barrier beach, Jetty Study investigations, and biological and physical monitoring of the Estuary. The purpose of Estuary management monitoring is to record any pinniped disturbances during the above activities.

A barrier beach was formed eleven times during 2015, but only during four of these closure events did the Water Agency artificially breach the sand bar. The Russian River mouth was closed to the ocean for a total of 115 days (or 32%) in 2015, mostly during the fall months. Pinniped monitoring occurred no more than 3 days before, the day of, and the day after each water level management activity.

The Water Agency's biological and physical monitoring activities of the Estuary are included in the NMFS IHA. The Water Agency surveys the sandbar (or barrier beach) monthly to collect a topographic map of the beach, as required by the Russian River Biological Opinion. A monitor is present during these surveys to record any disturbances of the Jenner haul-out during the survey. In 2015 the Water Agency completed the Jetty Study Plan (ESA PWA 2011) and a pinniped monitor was present to record any disturbances of the Jenner haul-out, similar to the monthly topographic surveys. Additionally, Water Agency field staff conducting biological and physical monitoring in the Estuary recorded any pinnipeds they encountered hauled out and any disturbance to pinnipeds associated with their activities.

The Estuary management and monitoring activities in 2015 resulted in incidental harassment (Level B harassment) of 2,383 harbor seals and 1 California sea lion, well under the total allowed by NMFS IHA. The Estuary management activities in 2014, 2013, 2012, 2011 and 2010 resulted in incidental harassment (Level B harassment) of 2,121, 1,351, 208, 42 and 290 harbor seals, respectively.

Jetty Study

The Russian River Biological Opinion, RPA 2, includes a step if adaptive management of the outlet channel as described, "is not able to reliably achieve the targeted annual and seasonal Estuary management water surface elevations by the end of 2010, Water Agency will draft a study plan for analyzing the effects and role of the Russian River jetty at Jenner on beach permeability, seasonal sand storage and transport, seasonal flood risk, and seasonal water surface elevations in the Estuary. That study will also evaluate alternatives for achieving targeted estuarine management water surface elevations via jetty removal, partial removal of the jetty, jetty notching, and potential use of the jetty as a tool in maintaining the estuary water surface elevations described above."

ESA PWA, at the request of the Water Agency, developed a plan to study the effects of the Goat Rock State Beach jetty on the Estuary in 2011 (ESA PWA 2011). In addition, it described the recommended approach for developing and assessing the feasibility of alternatives to the existing jetty that may help achieve target estuarine water surface elevations. As such, this study plan fulfills a portion of the Water Agency's obligations under the Biological Opinion. The Biological Opinion directs the Water Agency to change its management of the Estuary's water surface elevations with the intent of improving juvenile salmonid habitat while minimizing flood risk. Geophysical field studies were completed in 2014. The final report is currently being prepared and the report will be included in the next annual report.

Flood Risk Management

The Russian River Biological Opinion, RPA 2, includes a Flood Risk Reduction step if it proves difficult to reliably achieve raised water surface elevation targets based on implementation of a lagoon outlet channel or modification of the existing jetty. Should those actions be unsuccessful in meeting estuarine water surface elevation goals, RPA 2 states that the Water Agency "will evaluate, in coordination with NMFS and other appropriate public agencies, the feasibility of actions to avoid or mitigate damages to structures in the town of Jenner and low-lying properties along the Estuary that are currently threatened with flooding and prolonged inundation when the barrier beach closes and the Estuary's water surface elevation rises above 9 feet. Such actions may include, but are not limited to, elevating structures to avoid flooding or inundation."

The first effort to address flood risk management feasibility was compilation of a preliminary list of structures, properties, and infrastructure that would be subject to flooding/inundation as the result of sandbar formation and if the Estuary were allowed to naturally breach. As required by RPA 2, the Water Agency submitted a preliminary list of properties, structures, and infrastructure that may be subject to inundation if the barrier beach at the mouth of the Russian River was allowed to naturally breach. This preliminary list was updated for the California Coastal Commission Coastal Development Permit application process. Allowing Estuary water surface elevations to rise to between 10 and 12 feet NGVD (the estimated water surface elevation if the barrier beach was allowed to naturally breach per consultation with NMFS) may potentially inundate portions of properties.

The Water Agency is continuing to consult and coordinate with NMFS and the County of Sonoma's Local Coastal Plan update. The County's Permit Resources and Management Department is currently updating its Local Coastal Plan, including consideration of sea level rise impacts to the lower Russian River and community of Jenner. Updates to the Coastal Plan policies may result in additional evaluation of feasible engineering solutions to flood risk to low-lying properties along the Estuary. The Water Agency is participating, along with Sonoma County Permit and Resources Management Department, in NOAA's Habitat Blueprint, which includes a multiagency effort to develop and expand the United States Geological Survey (USGS) sea level rise model (the Coast Storm Modeling System or CoSMoS) to inform adaptation planning and Estuary management efforts. An updated flood risk report will be completed in 2016.

4.1 Water Quality Monitoring

Water quality monitoring was conducted in the lower, middle, and upper reaches of the Russian River Estuary, including two tributaries and the Maximum Backwater Area (MBA), between the mouth of the river at Jenner and Vacation Beach near Guerneville. Water Agency staff continued to collect data to establish baseline information on water quality in the Estuary, gain a better understanding of the longitudinal and vertical water quality profile during the ebb and flow of the tide, and track changes to the water quality profile that may occur during periods of barrier beach closure, partial or full lagoon formation, lagoon outlet channel implementation, and sandbar breach.

Saline water is denser than freshwater and a salinity “wedge” (halocline) forms in the Estuary as freshwater outflow passes over the denser tidal inflow. During the Lagoon Management Period, the lower and middle reaches of the Estuary up to Sheephouse Creek are predominantly saline environments with a thin freshwater layer that flows over the denser saltwater. The upper reach of the Estuary transitions to a predominantly freshwater environment, which is periodically underlain by a denser, saltwater layer that migrates upstream to Duncans Mills during summer low flow conditions and barrier beach closure. Additionally, river flows, tides, topography, and wind action affect the amount of mixing of the water column at various longitudinal and vertical positions within the reaches of the Estuary. The Maximum Backwater Area encompasses the area of the river between Duncans Mills and Vacation Beach that is generally outside the influence of saline water, but within the upper extent of inundation and backwatering that can occur during tidal cycles and lagoon formation.

Methods

Continuous Multi-Parameter Monitoring

Water quality was monitored using YSI Series 6600 multi-parameter datasondes. Hourly salinity (parts per thousand), water temperature (degrees Celsius), dissolved oxygen (percent saturation), dissolved oxygen (milligrams per liter), and pH (hydrogen ion) data were collected. Datasondes were cleaned and recalibrated periodically following the YSI User Manual procedures, and data was downloaded during each calibration event.

Nine stations were established for continuous water quality monitoring, including five stations in the mainstem Estuary, two tributary stations, and two stations in the MBA near Monte Rio (Figure 4.1.1). One mainstem Estuary station was located in the lower reach at the mouth of the Russian River at Goat Rock State Beach (Mouth Station). Two mainstem Estuary stations were placed in the middle reach: Patty’s Rock upstream of Penny Island (Patty’s Rock Station), and in the pool downstream of Sheephouse Creek (Sheephouse Creek Station). One tributary station was located in the mouth of Willow Creek, which flows into the middle reach of the Estuary (Willow Creek Station). Two mainstem Estuary stations were located in the upper reach; downstream of Freezeout Creek in Duncans Mills (Freezeout Creek Station) and downstream of Austin Creek in Brown’s Pool (Brown’s Pool Station). The other tributary station was located downstream of the first steel bridge in lower Austin Creek, which flows into the

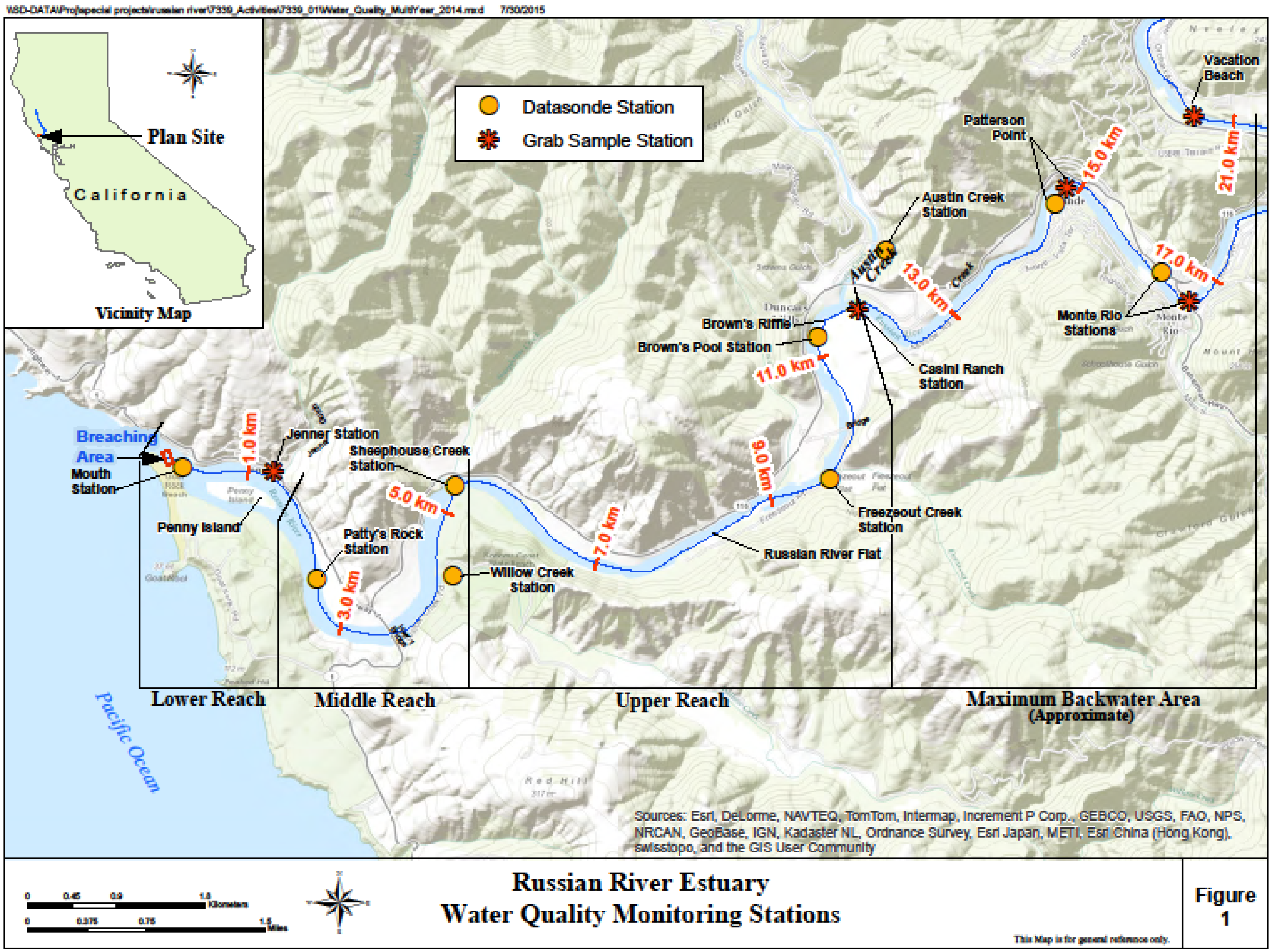


Figure 4.1.1. 2015 Russian River Estuary Water Quality Monitoring Stations

mainstem Russian River above Brown's Pool Station. Finally, two mainstem stations were located in the MBA; in a pool across from Patterson Point in Villa Grande (Patterson Point station) and downstream of Monte Rio Beach (Monte Rio Station).

The rationale for choosing mainstem Estuary sites, including the Brown's Pool Station, was to locate the deepest holes at various points throughout the Estuary to obtain the fullest vertical profiles possible and to monitor salinity circulation and stratification, hypoxic and/or anoxic events, and temperature stratification. Sondes were located near the mouths of Willow and Austin Creeks to collect baseline water quality conditions and monitor potential changes to water quality (e.g. salinity intrusion) resulting from tidal cycling or inundation during partial or full lagoon formation. The Patterson Point and Monte Rio stations were established to monitor potential changes to water quality conditions (including potential salinity migration) in the MBA while inundated during lagoon formation (Figure 4.1.1).

Mainstem Estuary and MBA monitoring stations up to Patterson Point were comprised of a concrete anchor attached to a steel cable suspended from the surface by a large buoy (Figure 4.1.2).

The Mouth, Patty's Rock, and Freezeout Creek stations had a vertical array of two datasondes to collect water quality profiles, whereas the Sheephouse Creek, Brown's Pool, and Patterson Point stations had one datasonde each. Stations in the lower and middle reaches of the Estuary that are predominantly saline had sondes placed at the surface, at approximately 1 meter depth (~1m), and/or at the mid-depth (~3m) portions of the water column. Stations in the upper reaches of the Estuary, where the halocline is deeper and the water is predominantly fresh to brackish, had sondes placed at the bottom (~6-8m) and/or mid-depth (~3-4m) portions of the water column. The Patterson Point monitoring station, located in the MBA, also had one datasonde placed at the bottom (~9-11m) of the pool (Figure 4.1.2). Sondes were located in this manner to track vertical and longitudinal changes in water quality characteristics during periods of tidal circulation, barrier beach closure, lagoon formation, lagoon outlet channel implementation, and sandbar breach.

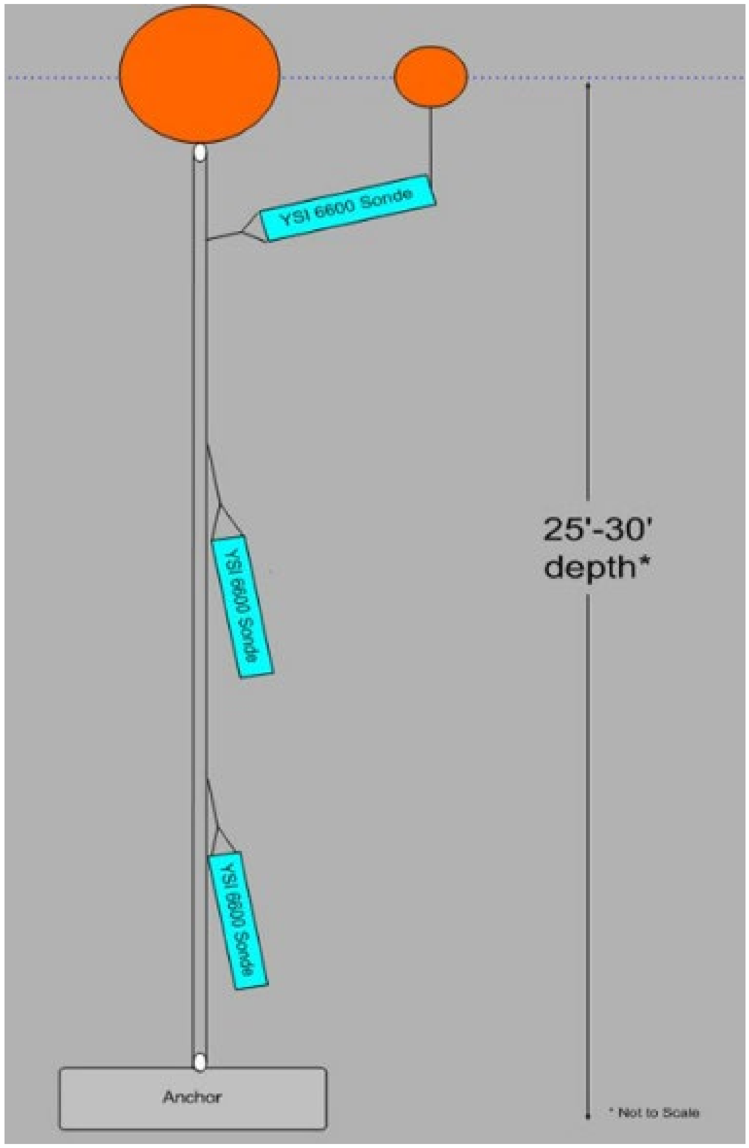


Figure 4.1.2. Typical Russian River Estuary monitoring station datasonde array.

The monitoring stations in Austin Creek, Willow Creek, and at Monte Rio consisted of one datasonde suspended at approximately mid-depth (~1m during open conditions) in the thalweg at each respective site.

Most of the stations were deployed from April through late November. The Mouth surface sonde and Monte Rio sonde were deployed in late May. The Austin Creek, Monte Rio, and Willow Creek sondes were deployed until December.

Grab Sample Collection

In 2015, Water Agency staff continued to conduct nutrient and indicator bacteria grab sampling at five stations in the Russian River Estuary and MBA, including three stations established in 2010: the Jenner Boat Ramp (Jenner Station); Casini Ranch across from the mouth of Austin Creek (Casini Ranch Station); and just downstream of the Monte Rio Bridge (Monte Rio

Station). The 2015 grab sampling effort represented the second year of collecting samples at Patterson Point in Villa Grande (Patterson Point Station); and just downstream of the Vacation Beach summer dam (Vacation Beach station). Refer to Figure 4.1.1 for grab sampling locations.

Water Agency staff collected grab samples weekly from May 12 to October 13. Additional focused sampling (collecting three samples over a ten-day period) was conducted following or during specific river management and operational events including: barrier beach closure, lagoon outlet channel implementation, sandbar breach, or removal of summer recreational dams. Additional bacterial sampling was also conducted when *Escherichia coli* (E. coli) conditions exceeded recommended criteria at a given station. Nutrient, chlorophyll a, and organic carbon grab samples were analyzed at Alpha Analytical Labs in Ukiah, and bacterial grab samples were analyzed at the Sonoma County Department of Health Services (DHS) lab in Santa Rosa.

Nutrient sampling was conducted for total organic nitrogen, ammonia, unionized ammonia, nitrate, nitrite, total Kjeldahl nitrogen, total nitrogen, and total phosphorus, as well as for chlorophyll a, which is a measurable parameter of algal growth that can be tied to excessive nutrient concentrations and reflect a biostimulatory response. Grab samples were collected for the presence of indicator bacteria including total coliforms, E. coli and Enterococcus. These bacteria are considered indicators of water quality conditions that may be a concern for water contact recreation and public health. The results of sampling conducted for total orthophosphate, dissolved organic carbon, total organic carbon, total dissolved solids, and turbidity are included as Appendix 4.5; however, an analysis and discussion of these constituents is not included in this report. Temperature, dissolved oxygen, pH, salinity, specific conductance, and turbidity values were recorded during grab sampling events and are included in Appendix 4.5.

Results

Water quality conditions in 2015 were similar to trends observed in sampling from 2004 to 2014, even with drought conditions and lower flows. The lower and middle reaches of the Estuary are predominantly saline environments with a thin freshwater layer that flows over the denser saltwater layer. The upper reach transitions to a predominantly freshwater environment, which is periodically underlain by a denser, saltwater layer that migrates up and downstream and appears to be affected in part by freshwater inflow rates, tidal inundation, barrier beach closure, and subsequent tidal cycles following reopening of the barrier beach. The river upstream of Brown's Pool is considered predominantly freshwater habitat. The lower and middle reaches of the Estuary are subject to tidally-influenced fluctuations in water depth during open conditions and inundation during barrier beach closure, as is the upper reach and the MBA to a lesser degree.

Table 4.1.1 presents a summary of minimum, mean, and maximum values for temperature, depth, dissolved oxygen (DO), pH, and salinity recorded at the various datasonde monitoring stations. Data associated with malfunctioning datasonde equipment has been removed from the data sets, resulting in the data gaps observed in the graphs presented as Figures 4.1.3 through 4.1.38. These data gaps may affect minimum, mean, and maximum values of the various constituents monitored in 2015, including temperature, dissolved oxygen, pH, and salinity at the Mouth mid-depth sonde in April, early May, and November, and the Freezeout Creek bottom sonde in September and early October. In addition, the Patterson Point station was removed between August and mid-September due to a lack of access associated with low summer flows.

Although gaps exist in the 2015 data that affect sample statistics, Water Agency staff has collected long time-series data on an hourly frequency for several years at most of these stations, and it is unlikely that the missing data appreciably affected the broader understanding of water quality conditions within the estuary. The following sections provide a brief discussion of the results observed for each parameter monitored.

Salinity

Full strength seawater has a salinity of approximately 35 parts per thousand (ppt), with salinity decreasing from the ocean to the upstream limit of the Estuary, which is considered freshwater at approximately 0.5 ppt (Horne 1994). All of the mid-depth sondes in the lower and middle reaches were located in a predominantly saline environment, whereas the surface sondes were located at the saltwater-freshwater interface (halocline or salt wedge) and recorded both freshwater and saltwater conditions. In the middle reach of the Estuary, salinities can range as high as 30 ppt in the saltwater layer, with brackish conditions prevailing at the upper end of the salt wedge, to less than 1 ppt in the freshwater layer on the surface. The Willow Creek sonde was located just upstream of the confluence with the Russian River, where predominantly freshwater conditions observed in the creek during higher springtime flows transitioned to a brackish environment during lower dry season flows.

In the upper reach, the Estuary typically transitions from predominantly saline conditions to brackish and freshwater conditions in the Heron Rookery area. Upstream, the Freezeout Creek station is located in a predominantly freshwater environment; however, brackish conditions can occur in the lower half of the water column during open estuary conditions with lower in-stream flows, as well as during barrier beach closure or perched conditions. The Brown’s Pool station is located in predominantly freshwater habitat in the upper reach of the Estuary, just downstream of the confluence with Austin Creek and the beginning of the MBA; however, brackish water was observed to occur at the bottom of the pool periodically through the 2015 monitoring season and at mid-depth during a closure in late October.

The Austin Creek, Patterson Point and Monte Rio stations are located in the MBA in freshwater habitat that can become inundated during high tides, barrier beach closures, perched conditions, and lagoon formation. Elevated salinity levels were not observed at any of the stations in the MBA during either open river mouth or closed barrier beach conditions in 2015.

Table 4.1.1. Russian River Estuary 2015 Water Quality Monitoring Results. Minimum, mean, and maximum values for temperature (degrees Celsius), depth (meters), dissolved oxygen (percent) saturation, dissolved oxygen concentration (milligrams per Liter), hydrogen ion (pH units), and salinity (parts per thousand).

Monitoring Station	**Temperature**	**Depth**	**Dissolved Oxygen**	**Dissolved Oxygen**	**Hydrogen Ion**	**Salinity**
Sonde	(°C)	(m)	(%) saturation	(mg/L)	(pH)	(ppt)
Mouth						
Surface						
April 22, 2015 - November 24, 2015						
Min	9.6	0.9	4.8	51.9	7.4	0.4
Mean	16.0	1.2	9.5	107.8	8.3	19.2
Max	**22.3**	**1.9**	**27.4**	**316.4**	**9.5**	**34.0**
Mid-Depth						
May 20, 2015 - November 12, 2015						
Min	10.3	3.0	0.1	0.6	7.1	6.1
Mean	15.5	3.4	7.2	85.2	7.8	29.0
Max	**21.5**	**3.8**	**15.2**	**176.5**	**8.4**	**34.0**
Patty's Rock						
Surface						
April 29, 2015 - November 24, 2015						
Min	9.5	0.6	5.0	62.6	7.2	0.3
Mean	17.0	0.8	9.6	109.5	8.0	15.5
Max	**22.5**	**1.1**	**17.0**	**211.5**	**9.4**	**33.1**
Mid-Depth						
April 29, 2015 - November 24, 2015						
Min	10.7	2.9	0.1	1.4	7.1	4.0
Mean	15.8	3.2	8.1	95.5	8.1	28.4
Max	**22.6**	**3.5**	**26.1**	**318.3**	**9.0**	**33.4**
Willow Creek						
Mid-Depth						
April 15, 2015 - December 15, 2015						
Min	6.8	0.2	0.0	0.0	6.4	0.1
Mean	18.7	1.0	6.5	74.3	7.6	11.7
Max	**25.7**	**3.6**	**18.8**	**237.7**	**8.8**	**27.8**
Sheephouse Creek						
Mid-Depth						
April 29, 2015 - November 24, 2015						
Min	12.2	3.3	0.1	0.7	7.0	1.3
Mean	17.6	3.5	7.0	85.4	7.8	26.8
Max	**22.0**	**3.7**	**14.2**	**172.2**	**8.5**	**32.1**
Freezeout Creek						
Mid-Depth						
April 29, 2015 - November 24, 2015						
Min	11.1	2.6	0.1	1.0	7.0	0.1
Mean	21.5	3.5	7.1	80.3	7.9	4.8
Max	**26.2**	**5.3**	**16.9**	**215.3**	**8.9**	**21.0**
Bottom						
April 22, 2015 - November 24, 2015						
Min	14.6	4.1	0.0	0.2	6.4	0.1
Mean	21.4	6.1	5.0	56.9	7.5	5.2
Max	**24.4**	**8.0**	**13.5**	**155.2**	**8.9**	**21.1**

(continues on next page)

Table 4.1.1 (cont.). Russian River Estuary 2015 Water Quality Monitoring Results. Minimum, mean, and maximum values for temperature (degrees Celsius), depth (meters), dissolved oxygen (percent) saturation, dissolved oxygen concentration (milligrams per Liter), hydrogen ion (pH units), and salinity (parts per thousand).

Monitoring Station	**Temperature**	**Depth**	**Dissolved Oxygen**	**Dissolved Oxygen**	**Hydrogen Ion**	**Salinity**
Sonde	(°C)	(m)	(%) saturation	(mg/L)	(pH)	(ppt)
Brown's Pool						
Mid-Depth						
April 21, 2015 - November 23, 2015						
Min	10.8	4.8	0.1	1.1	7.1	0.1
Mean	20.3	5.2	7.9	86.7	8.0	0.7
Max	**24.8**	**5.7**	**12.2**	**139.2**	**8.9**	**9.5**
Bottom						
April 21, 2015 - November 23, 2015						
Min	10.7	8.6	0.1	0.6	6.2	0.1
Mean	18.8	9.8	2.8	30.2	7.2	1.3
Max	**23.9**	**10.5**	**10.2**	**116.0**	**8.3**	**9.7**
Austin Creek						
Mid-Depth						
April 13, 2015 - December 31, 2015						
Min	7.3	0.0	0.1	0.9	6.9	0.0
Mean	15.8	0.6	5.0	49.4	7.4	0.2
Max	**22.3**	**2.9**	**11.1**	**114.5**	**8.0**	**0.2**
Patterson Point						
Bottom						
April 7, 2015 - November 23, 2015						
Min	10.0	9.6	0.1	0.8	6.3	0.1
Mean	17.4	10.2	4.1	41.4	7.2	0.2
Max	**20.7**	**11.3**	**10.6**	**109.1**	**8.2**	**0.4**
Monte Rio						
Mid-Depth						
May 21, 2015 - December 2, 2015						
Min	7.0	0.6	6.2	70.9	7.5	0.1
Mean	20.5	1.0	8.6	94.6	7.9	0.1
Max	**27.1**	**1.9**	**12.1**	**118.1**	**8.5**	**0.2**

Lower and Middle Reach Salinity

The surface sondes at the Mouth and Patty's Rock stations were suspended at a depth of approximately 1 meter, and experienced frequent hourly fluctuations in salinity during open conditions. These fluctuations are influenced by freshwater inflows, tidal movement and expansion and contraction of the salt wedge. The freshwater layer was observed to be more persistent at the surface sondes during closed barrier beach conditions in the spring and fall (Figures 4.1.3 and 4.1.4). Concentrations ranged from 0.4 to 34.0 ppt at the Mouth surface sonde and 0.3 to 33.1 ppt at the Patty's Rock surface sonde (Table 4.1.1). The surface sondes at the Mouth and Patty's Rock had mean salinity values of 19.2 and 15.5 ppt, respectively.

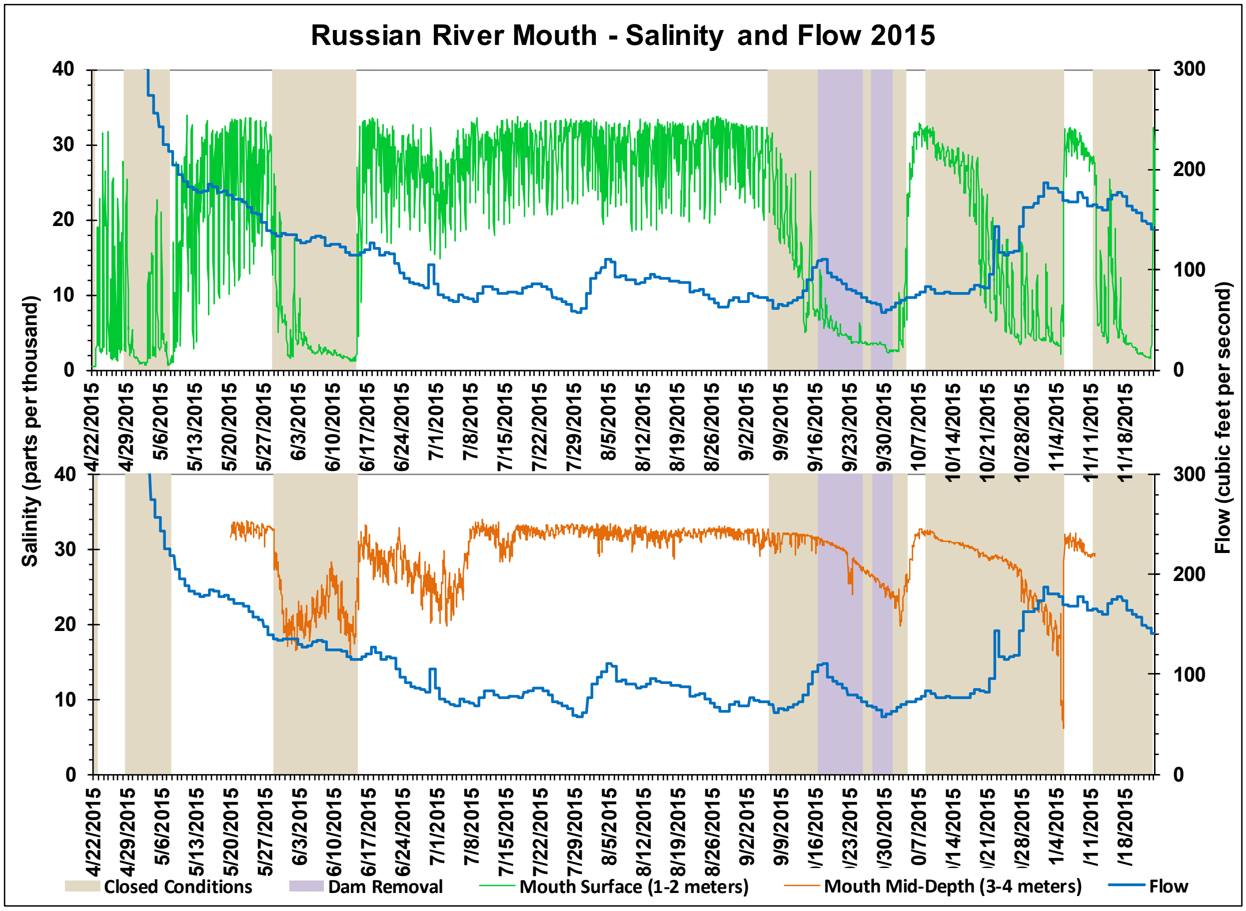


Figure 4.1.3. 2015 Russian River Mouth Salinity and Flow Graph.

The mid-depth sondes at the Mouth, Patty's Rock, and Sheephouse Creek stations were suspended at a depth of approximately 3 meters, and also experienced frequent fluctuations in salinity during open conditions, though to a lesser degree than their respective surface sondes. Concentrations ranged from 6.1 to 34.0 ppt at the Mouth, 4.0 to 33.4 ppt at Patty's Rock, and 1.3 to 32.1 ppt at Sheephouse Creek (Table 4.1.1). The mid-depth sondes at the Mouth, Patty's Rock, and Sheephouse Creek had mean salinity values of 29.0, 28.4, and 26.8 ppt, respectively. Minimum concentrations were observed to occur at the Mouth and Patty's Rock mid-depth sondes in November shortly after the barrier beach was breached by Water Agency staff (Figures 4.1.3 and 4.1.4). Minimum concentrations at Sheephouse Creek were also observed to occur after the breaching of the barrier beach in November (Figure 4.1.5).

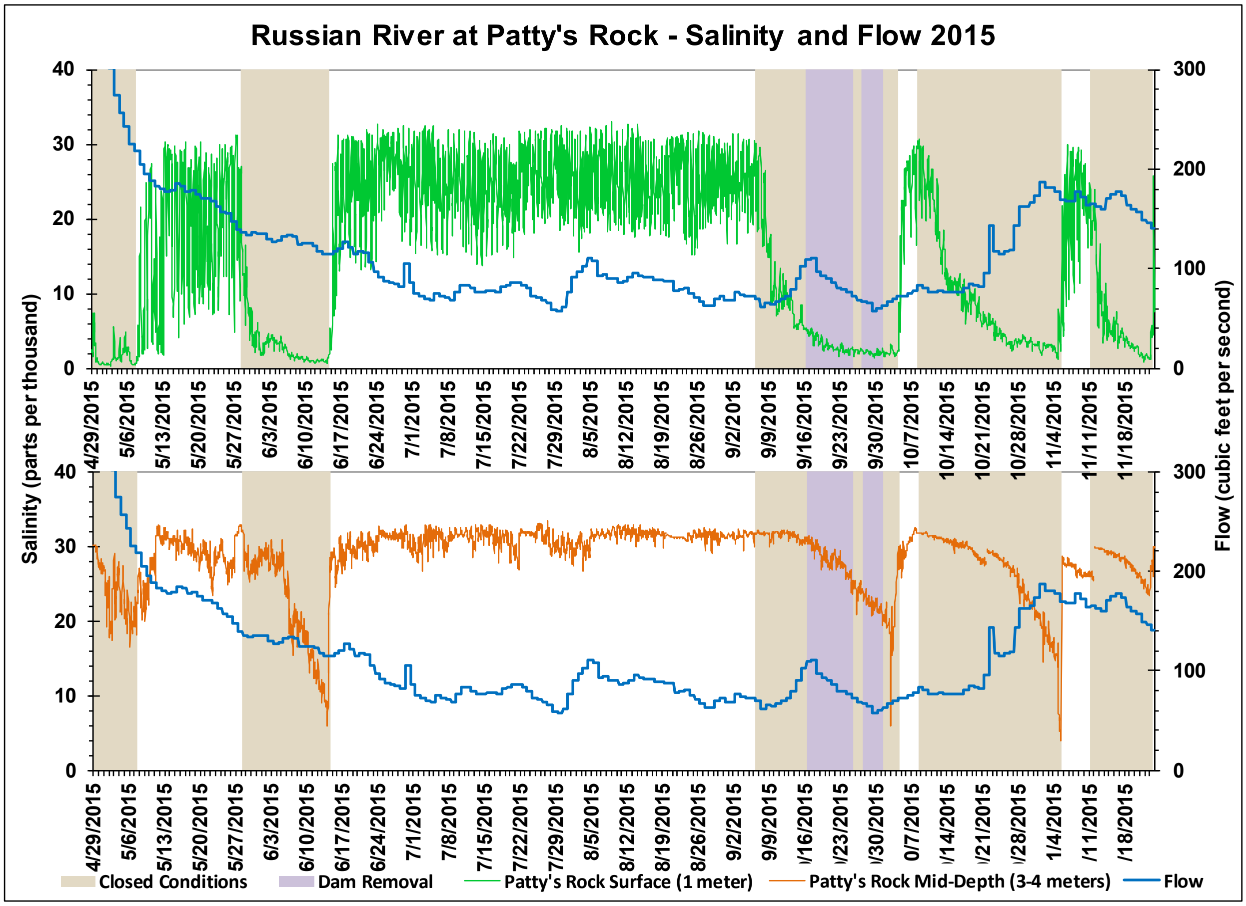


Figure 4.1.4. 2015 Russian River at Patty's Rock Salinity and Flow Graph

The Estuary experienced three closures during the 2015 management period, including a closure that lasted 28 days from 7 September to 4 October before opening naturally (Figure 4.1.6). Declines in salinity during barrier beach closure and lagoon formation were due to a combination of freshwater inflows increasing the depth of the freshwater layer over the salt layer, a reduction in tidal inflow, the compression and leveling out of the salt layer, and seepage of saline water through the barrier beach. Salinity generally returned to pre-closure levels after the barrier beach reopened, although the time required to return to pre-closure conditions varied at each site and differed between closure events. This variability was related to the strength of subsequent tidal cycles, freshwater inflow rates, topography, relative location within the Estuary, and to a lesser degree, wind mixing.

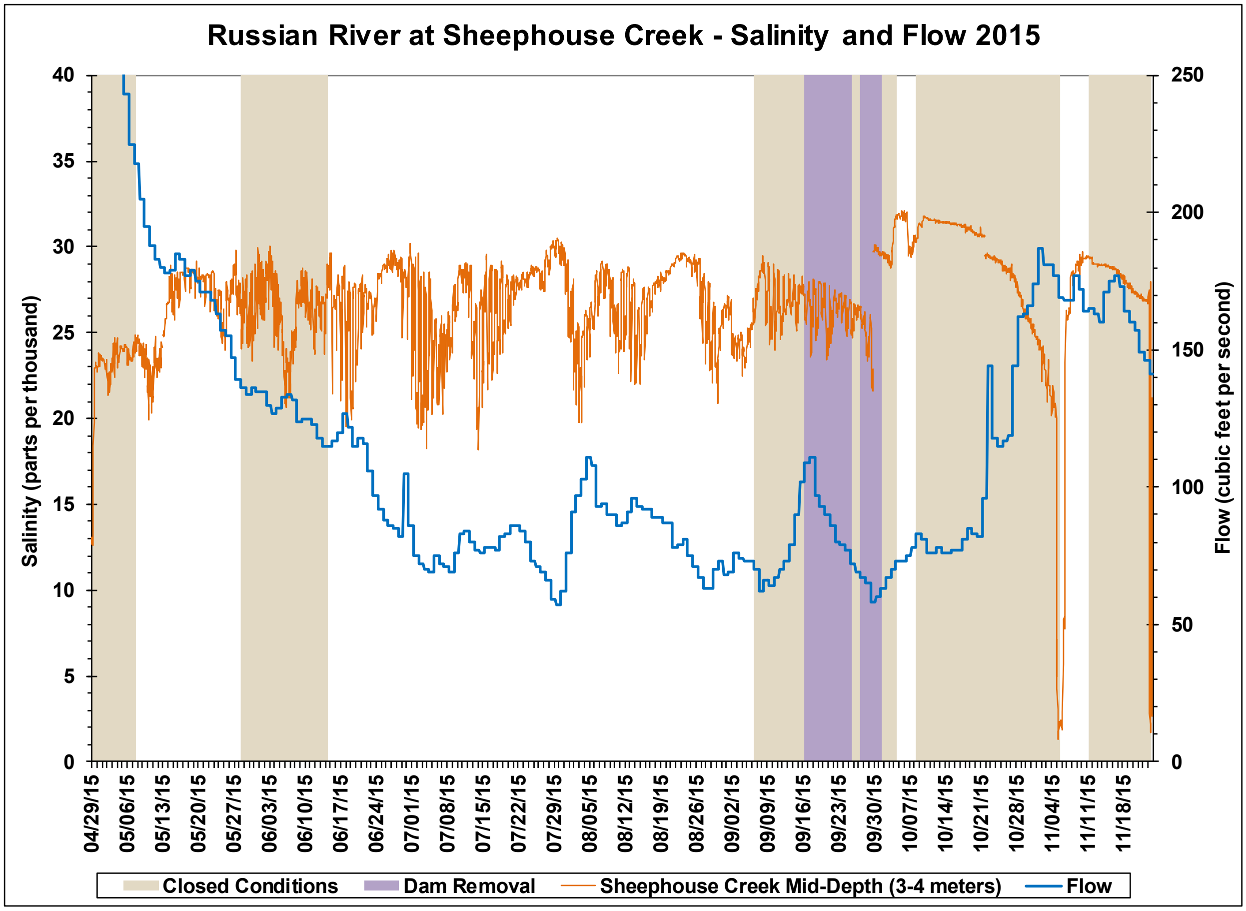


Figure 4.1.5. 2015 Russian River at Sheephouse Creek Salinity and Flow Graph

Figure 4.1.6. Russian River Mouth and Jetty from Jenner Overlook – September 8, 2015

The Willow Creek station was located in predominantly freshwater conditions through mid-May until spring flows receded below 200 cfs in the mainstem Russian River and increased tidal action allowed saline water to migrate to this station. Salinity was observed to decline during the extended closure in late-May, but remained brackish through the rest of the monitoring season, including during late season closures (Figure 4.1.7). Salinity was observed to decrease following the opening of the barrier beach late in the season, however, brackish conditions generally returned within a few days.

Salinity concentrations fluctuated significantly during open conditions with concentrations that ranged between 6 and 26 ppt from mid-May to early September. Salinity concentrations became more stable during the barrier beach closures in September and October and were observed to slowly decline through the closures. The mean salinity concentration observed at the Willow Creek station was 11.7 ppt, with a minimum concentration of 0.1 ppt, and a maximum concentration of 27.8 ppt (Table 4.1.1).

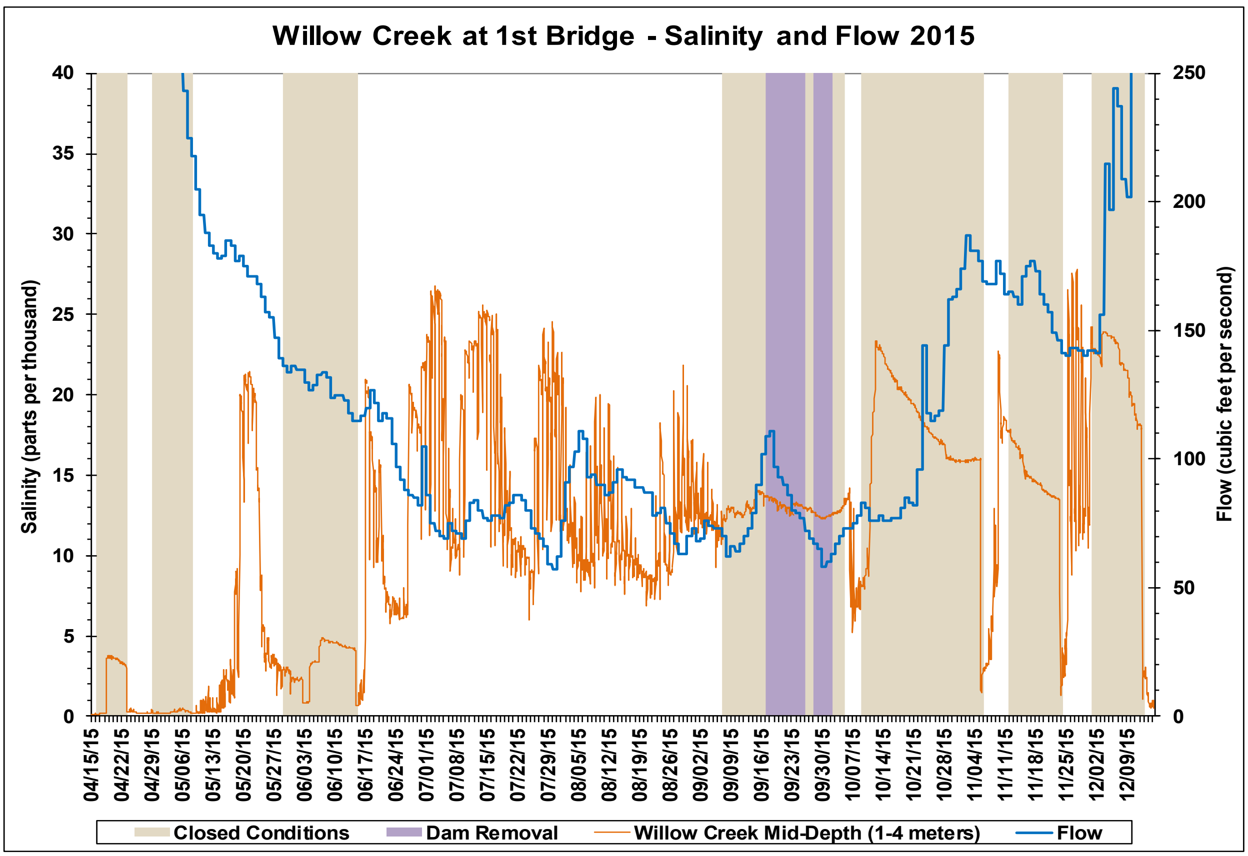


Figure 4.1.7. 2015 Willow Creek Salinity and Russian River Flow Graph

Upper Reach Salinity

Two stations were monitored in the upper reach in 2015; Freezeout Creek and Brown's Pool. Both stations included a bottom sonde and a mid-depth sonde. Sondes were located in this manner to track changes in the presence and concentration of salinity in the water column as well as the presence of thermal refugia for salmonids.

The Freezeout Creek station is located at River Kilometer 9.5 (RK 9.5), which is approximately 9.5 km upstream from the river mouth, in a pool approximately 300 meters downstream of the confluence of Freezeout Creek and the mainstem of the river. This station was located in a predominantly freshwater condition that was subject to elevated salinity levels as the salt wedge migrated up the Estuary during both open and closed conditions (Figure 4.1.8). The elevated salinity levels were predominantly observed at the bottom sonde, though elevated salinity was also seen at the mid-depth sonde during open and closed conditions. The bottom sonde at Freezeout Creek had a mean salinity concentration of 5.2 ppt, and salinity levels that ranged from 0.1 to 21.1 ppt (Table 4.1.1). The mid-depth sonde at Freezeout Creek had a mean salinity concentration of 4.8 ppt, and salinity levels that ranged from 0.1 to 21.0 ppt (Table 4.1.1).

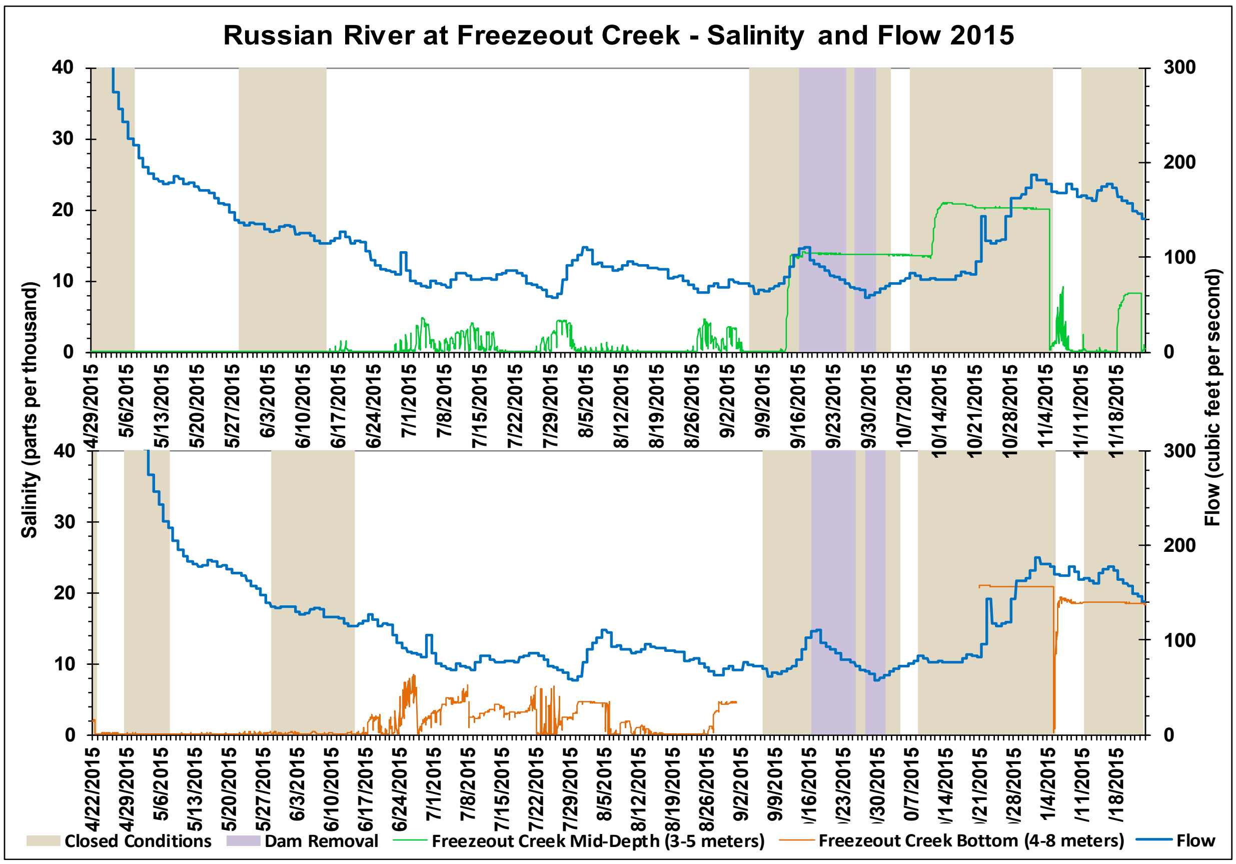


Figure 4.1.8. 2015 Russian River at Freezeout Creek Salinity and Flow Graph

The Brown's Pool station is located at RK 11.3 in a pool that is approximately 10m deep. Brown's Pool is located immediately downstream of Brown's Riffle (RK 11.4) and the confluence of Austin Creek and the mainstem Russian River, which is located at RK 11.65. Brown's Riffle is generally considered the demarcation between the Estuary and the MBA, where salinity levels have not been observed to occur past this point. The Brown's Pool bottom and mid-depth sondes were observed to remain a predominantly freshwater habitat during the 2015 monitoring season under open and closed conditions, with a few exceptions (Figure 4.1.8).

During the barrier beach closure in October, salinity concentrations at Brown's Pool were observed to increase to approximately 10 ppt at the mid-depth and bottom sondes by 30

October. Salinity concentrations were observed to decrease to freshwater conditions at the mid-depth sonde after the barrier beach was opened on 5 November. Salinity also briefly decreased at the bottom sonde before returning to brackish conditions, which persisted into the next closure until being replaced by freshwater on 18 November (Figure 4.1.9). The bottom sonde at Brown's Pool had a mean salinity concentration of 1.3 ppt, and salinity levels that ranged from 0.1 to 9.7 ppt (Table 4.1.1). The mid-depth sonde at Brown's Pool had a mean salinity concentration of 0.7 ppt, and salinity levels that ranged from 0.1 to 9.5 ppt (Table 4.1.1).

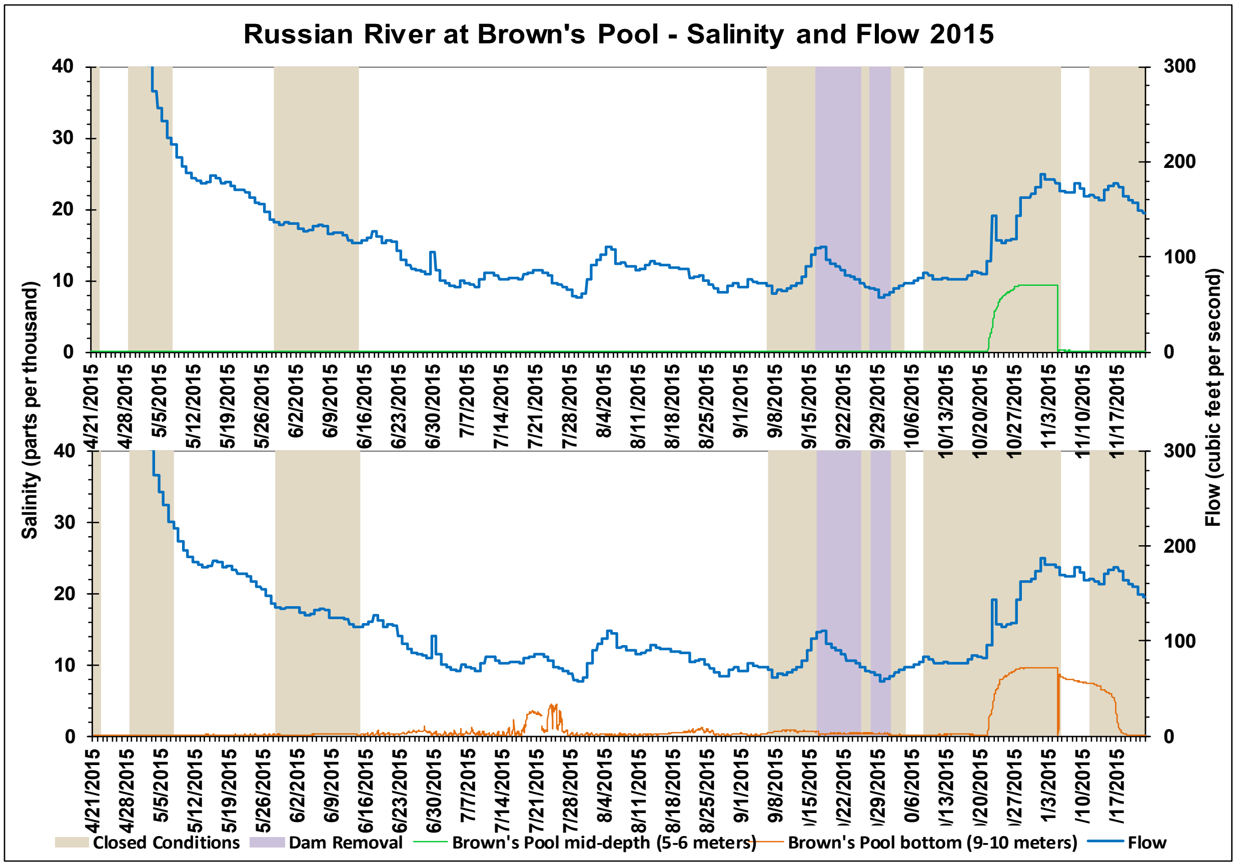


Figure 4.1.9. 2015 Russian River at Brown's Pool Salinity and Flow Graph

Maximum Backwater Area Salinity

Three stations were located in the MBA, including one tributary station in lower Austin Creek and two mainstem Russian River stations located in Patterson Point (RK 14.9) and Monte Rio (RK 16.1) (Figure 4.1.1). None of these three stations were observed to have salinity levels above normal background conditions expected in freshwater habitats, during both open and closed barrier beach conditions (Figures 4.1.10 through 4.1.12).

The Austin Creek station had a mean salinity concentration of 0.2 ppt, with a minimum of 0.0 ppt and a maximum of 0.2 ppt. The Patterson Point station had a mean salinity concentration of 0.2 ppt, a minimum concentration of 0.1 ppt, and a maximum concentration of 0.4 ppt. The Monte Rio station had a mean salinity concentration of 0.1 ppt, a minimum concentration of 0.1 ppt, and a maximum concentration of 0.2 ppt.

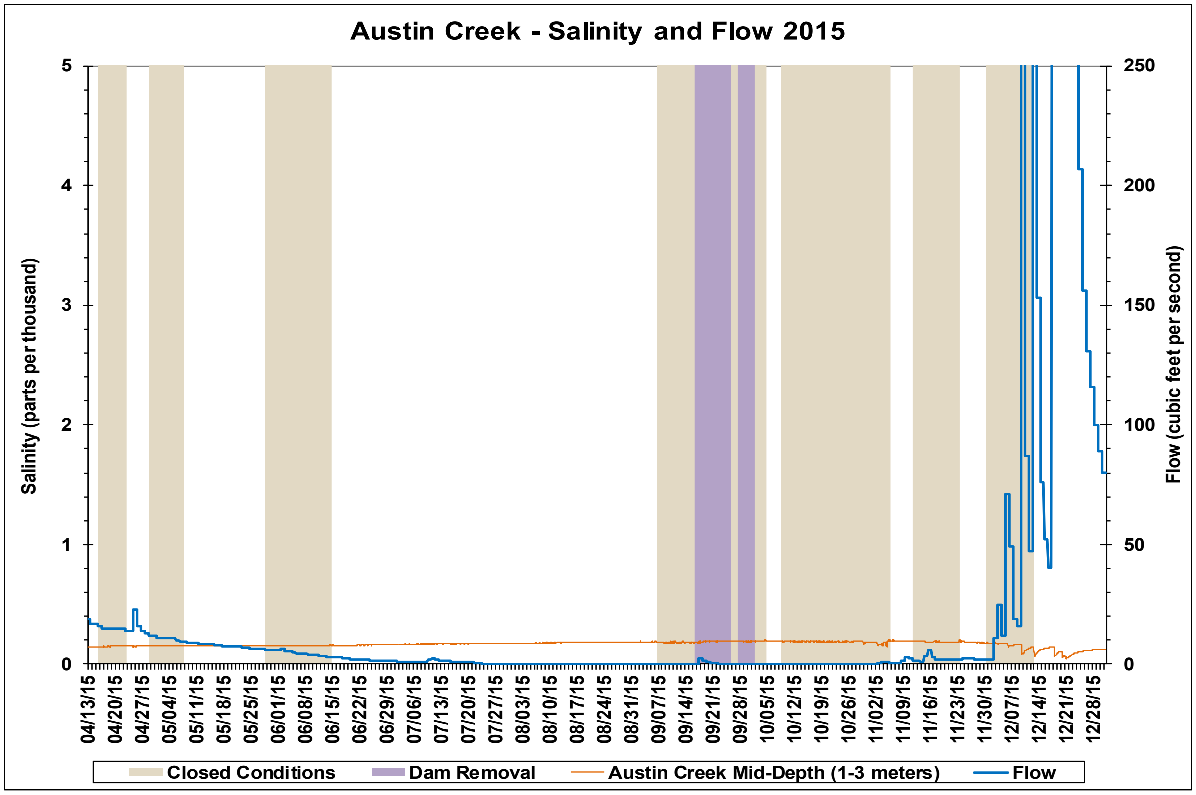


Figure 4.1.10. 2015 Austin Creek Salinity and Flow Graph

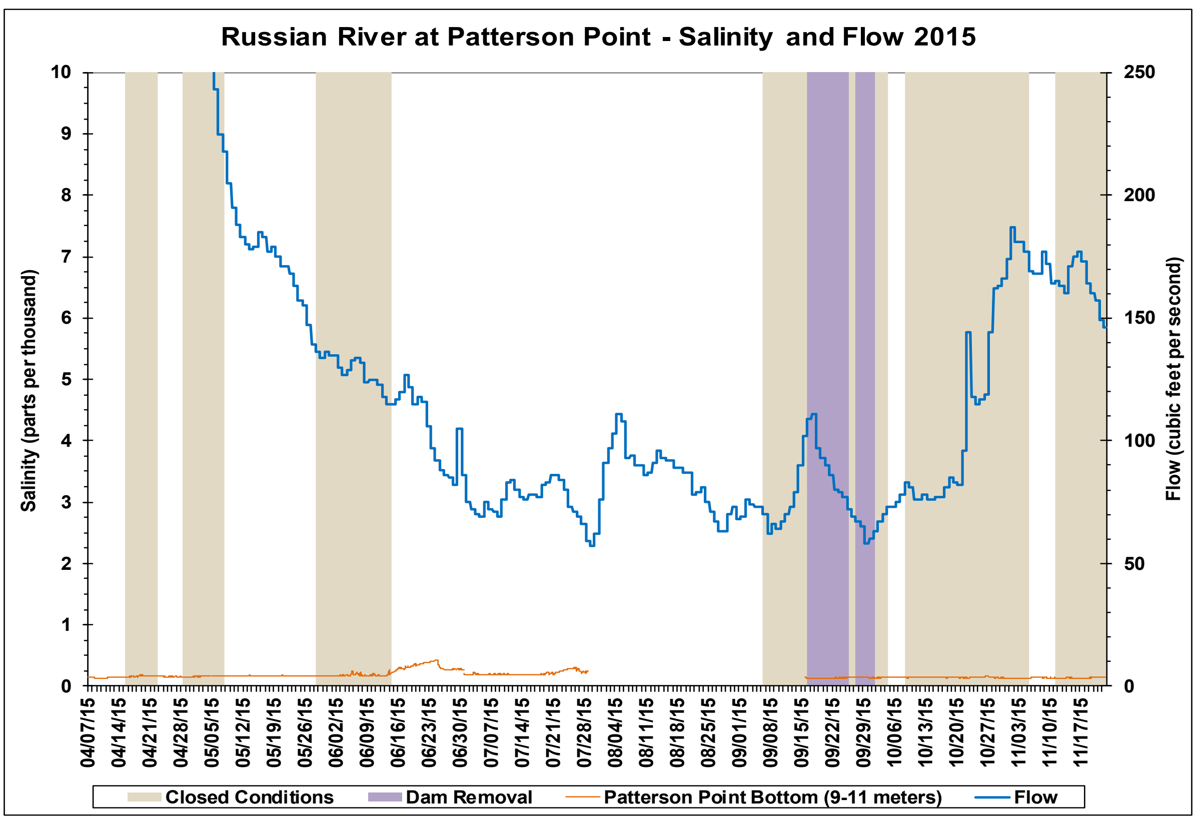


Figure 4.1.11. 2015 Patterson Point Salinity and Flow Graph

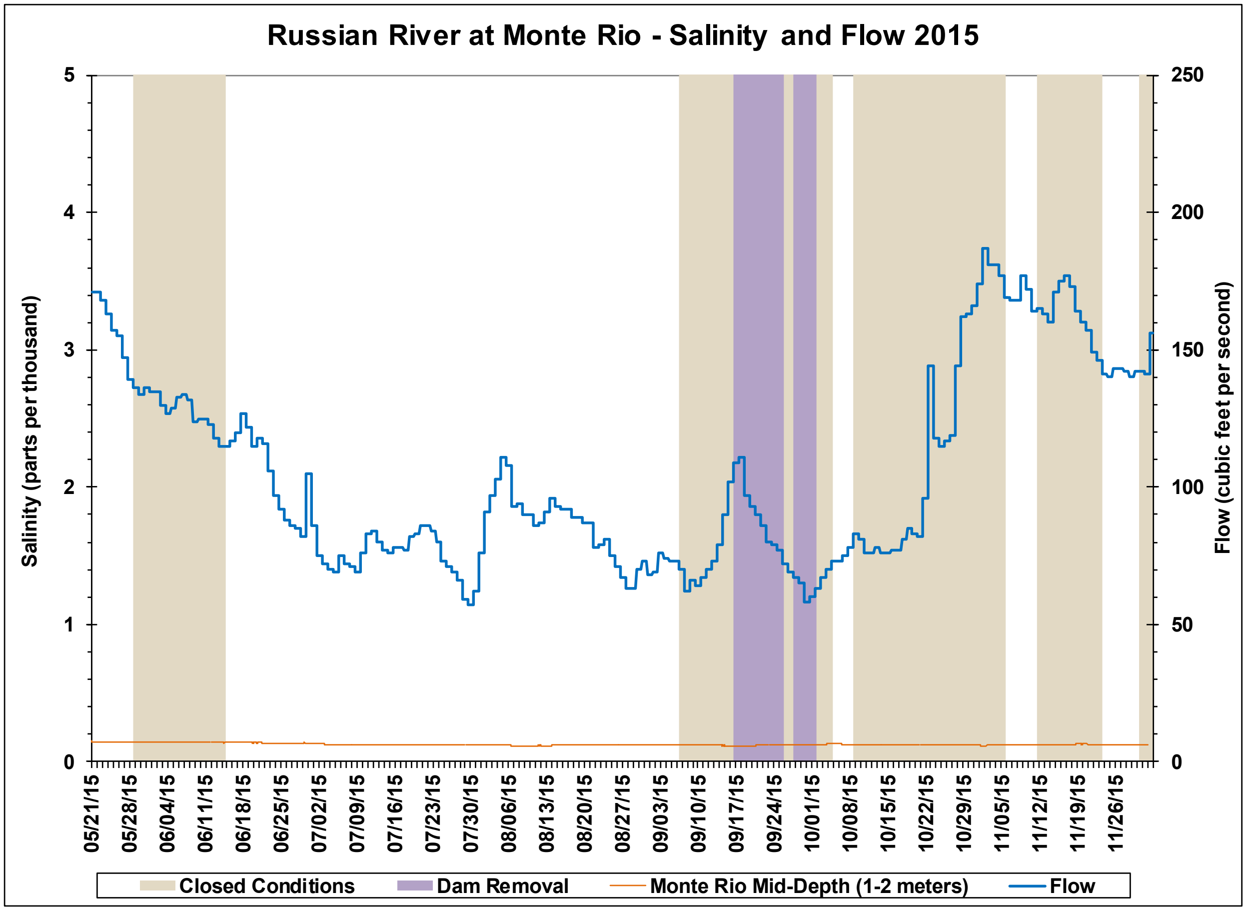


Figure 4.1.12. 2015 Russian River at Monte Rio Salinity and Flow Graph

Temperature

During open estuary conditions, mainstem water temperatures were reflective of the halocline, with lower mean and maximum temperatures typically being observed in the saline layer at the bottom and mid-depth sondes compared to temperatures recorded in the freshwater layer at the mid-depth and surface sondes (Figures 4.1.12 through 4.1.20). The differences in temperatures between the underlying saline layer and the overlying freshwater layer can be attributed in part to the source of saline and fresh water. During open estuary conditions, the Pacific Ocean, where temperatures are typically around 10 degrees Celsius (°C), is the source of saltwater in the Estuary. Whereas, the mainstem Russian River, with water temperatures reaching as high as 27 °C in the interior valleys, is the primary source of freshwater in the Estuary.

During closed Estuary conditions, increasing temperatures associated with fresh/saltwater stratification were observed to occur (Figures 4.1.13 through 4.1.15). Density and temperature gradients between freshwater and saltwater play a role in stratification and serve to prevent/minimize mixing of the freshwater and saline layers. When the estuary is closed, or the river mouth is perched and the supply of cool tidal inflow is reduced, solar radiation heats the underlying saline layer. Additionally, the overlying freshwater surface layer restricts the release of this heat, which can result in higher water temperatures in the underlying saline layer than in the overlying freshwater layer (Figures 4.1.13 and 4.1.14). Stratification based heating has also been observed to result in higher temperatures in the mid-depth saline layer compared to the bottom layer in deep pools, forming a three layered system. This stratification based heating can also contribute to higher seasonal mean temperatures in the saline layer than would be expected to occur under open conditions.

Lower and Middle Reach Temperature

The surface sondes were located at the freshwater/saltwater interface and were observed to have maximum temperatures of 22.3 and 22.5 °C at the Mouth and Patty's Rock, respectively. Whereas, the mid-depth sondes were located primarily in saltwater and had maximum temperatures of 21.5, 22.6, and 22.0 °C at the Mouth, Patty's Rock, and Sheephouse Creek, respectively (Table 4.1.1). The surface sondes had mean temperatures of 16.0 and 17.0 °C and minimum temperatures of 9.6 and 9.5 °C at the Mouth and Patty's Rock, respectively (Table 4.1.1). The mid-depth sondes had mean temperatures of 15.5, 15.8, and 17.6 °C, and minimum temperatures of 10.3, 10.7, and 12.2 °C at the Mouth, Patty's Rock, and Sheephouse Creek, respectively (Table 4.1.1).

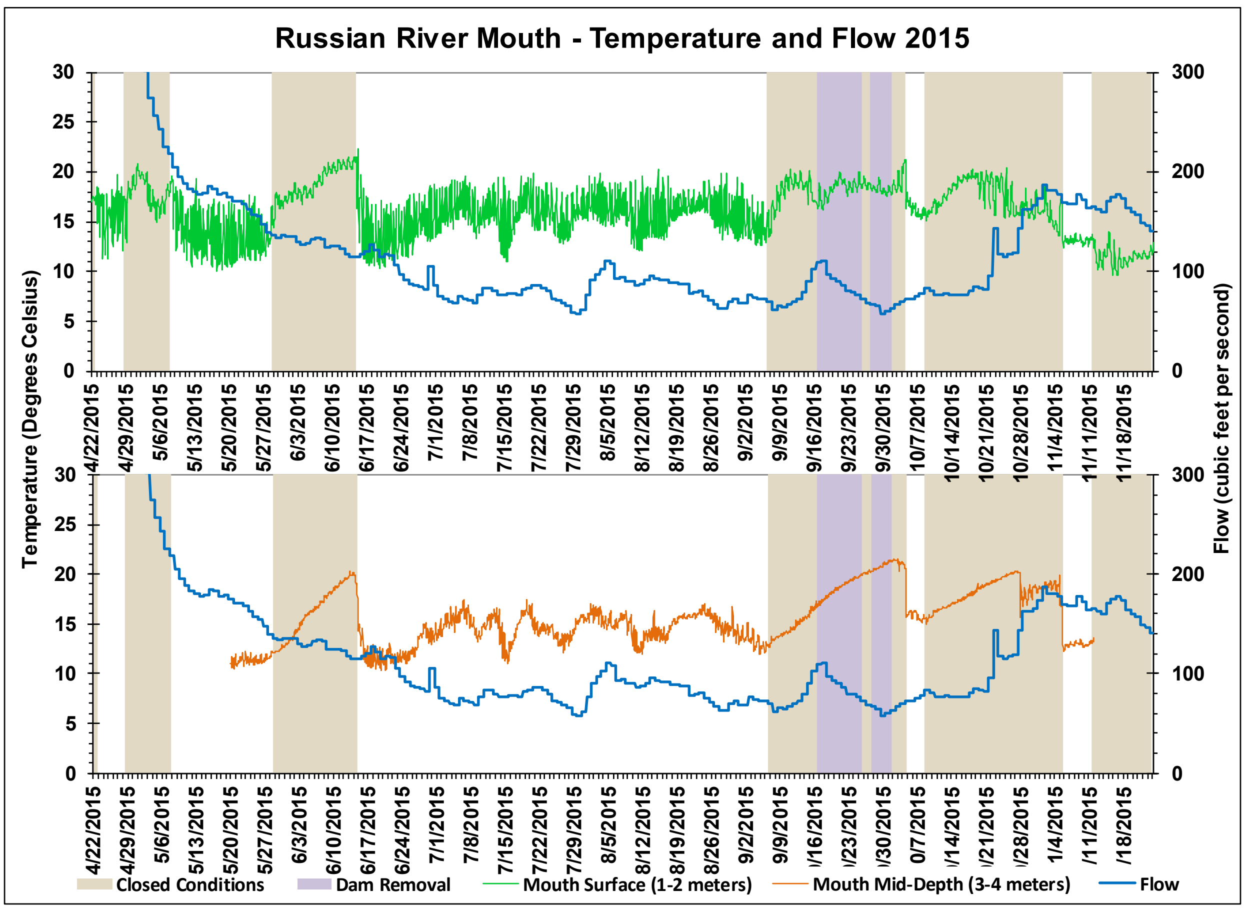


Figure 4.1.13. 2015 Russian River Mouth Temperature and Flow Graph

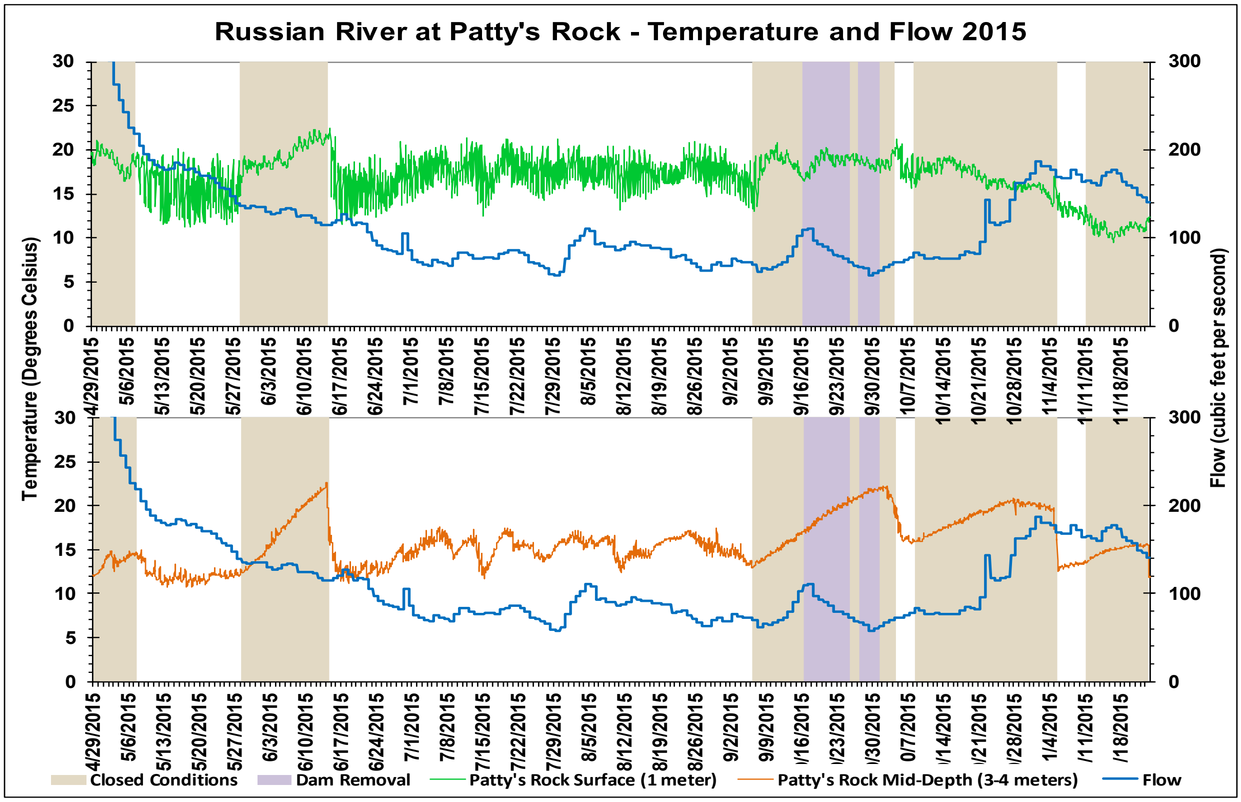


Figure 4.1.14. 2015 Russian River at Patty's Rock Temperature and Flow Graph

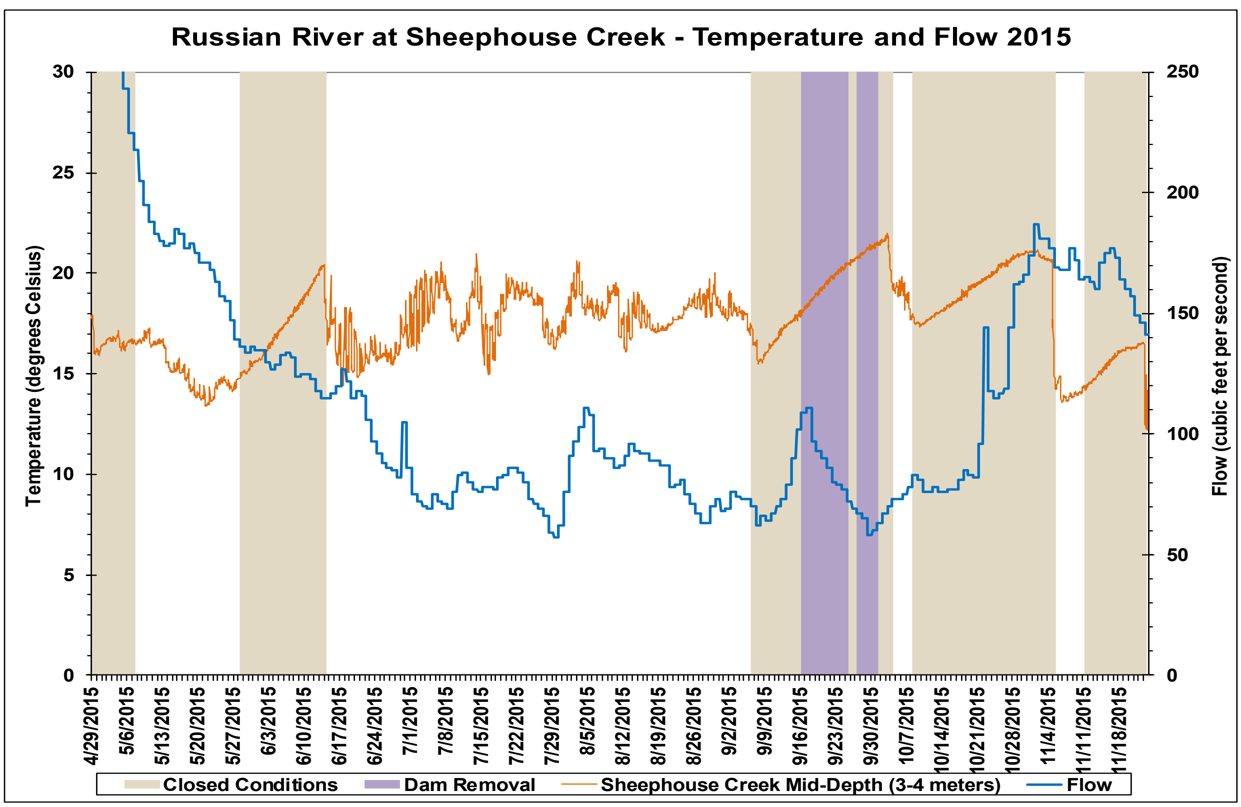


Figure 4.1.15. 2015 Russian River at Sheephouse Creek Temperature and Flow Graph

The Willow Creek station had a maximum temperature of 25.7 °C, which occurred on 20 July in brackish water and open conditions (Figures 4.1.16 and 4.1.7). The mean temperature was 18.7 °C, and the minimum temperature was 6.8 °C. Willow Creek had freshwater conditions prior to the monitoring season that became brackish to saline as flows dropped below 200 cfs in early May (Figure 4.1.7). The station remained brackish through early summer with periodic fluctuations as saline water migrated up and down stream with the tides. Temperatures were observed to fluctuate with the movement of saline water into and out of the station, resulting in both heating and cooling during open and closed Estuary conditions (Figure 4.1.16). This was most apparent during several late season barrier beach closure events when warm brackish water was observed to significantly decrease in temperature after freshwater and/or a fresh source of tidally migrating water migrated to the station during and between barrier beach closures (Figure 4.1.16).

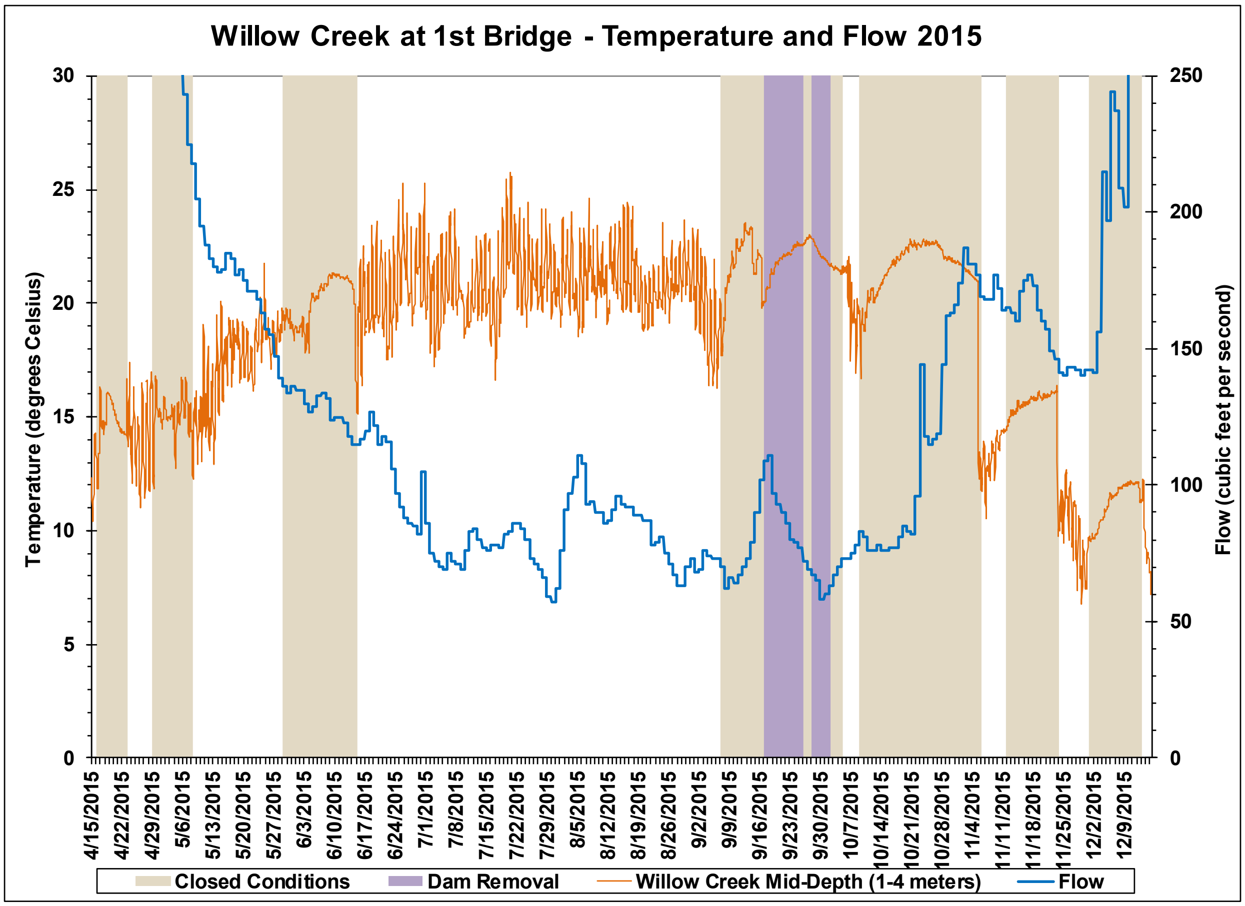


Figure 4.1.16. 2015 Willow Creek Temperature with Russian River Flow

Upper Reach Temperature

Overall estuarine temperatures in both the saline layer and freshwater layer were typically hottest at the upper reach stations, as observed at Freezeout Creek and Brown's Pool, and became progressively cooler as the water flowed downstream, closer to the cooling effects of the coast and ocean.

The bottom sonde at the Freezeout Creek station had a maximum temperature of 24.4 °C, a mean temperature of 21.4 °C, and a minimum temperature of 14.6 °C (Table 4.1.1). The mid-depth sonde had a maximum temperature of 26.2 °C, a mean temperature of 21.5 °C, and a minimum temperature of 11.1 °C. Minimum temperatures at the mid-depth sonde occurred in freshwater during closed conditions in November (Figure 4.1.17). Minimum temperatures at the bottom sonde occurred when freshwater briefly replaced warmer brackish water after Water Agency staff breached the barrier beach on 5 November (Figure 4.1.17). The maximum temperatures were observed to occur in predominantly freshwater conditions during open estuary conditions in late June at the bottom sonde and in late July at the mid-depth sonde. However, temperatures were also elevated at the mid-depth sonde and near the seasonal maximum value in brackish water during closed conditions in October. (Figure 4.1.17). Temperatures were observed to be fairly stable in the brackish layer during closures later in the season and were observed to decrease at the mid-depth sonde between November closures as freshwater replaced and/or mixed with the brackish layer (Figures 4.1.8 and 4.1.17).

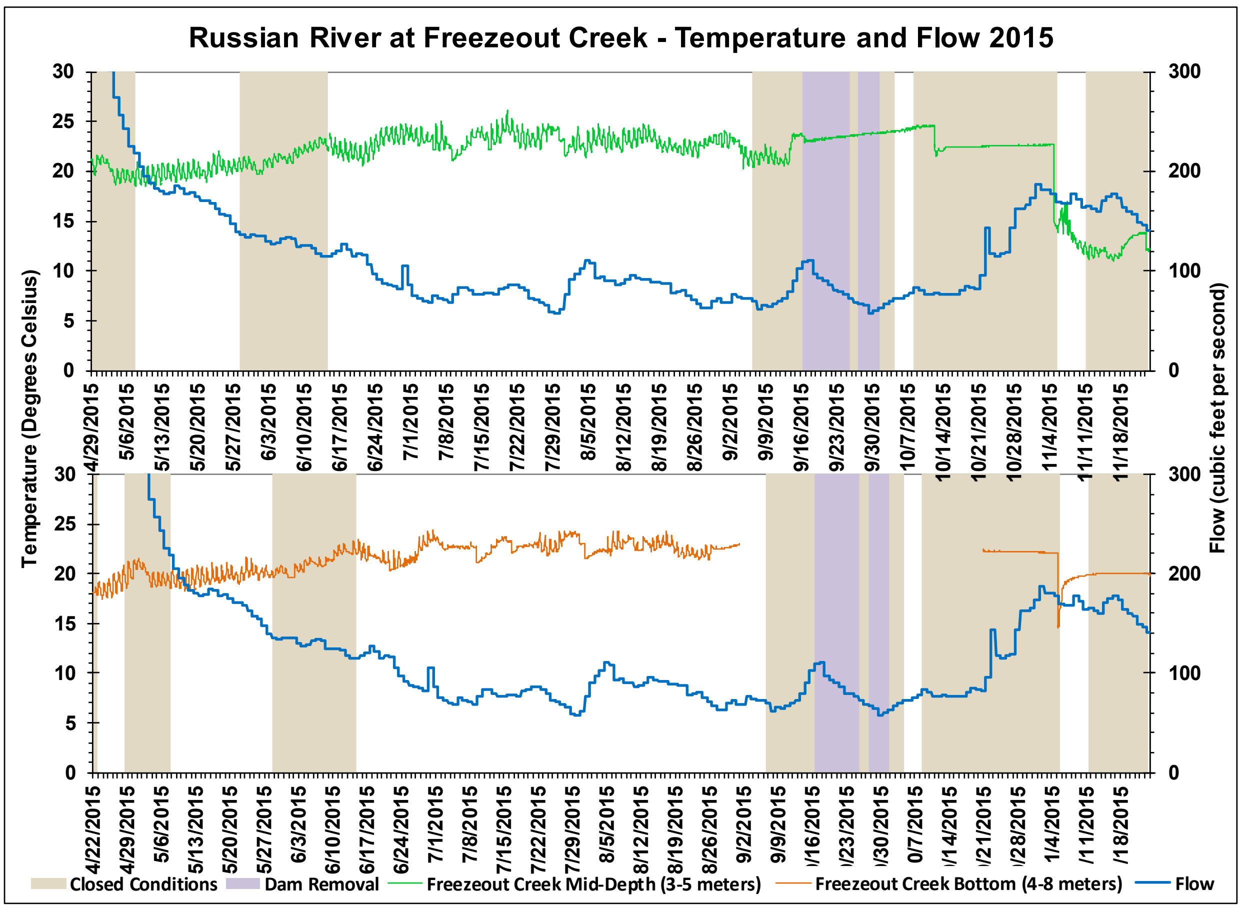


Figure 4.1.17. 2015 Russian River at Freezeout Creek Temperature and Flow Graph

The bottom sonde at the Brown's Pool station had a maximum temperature of 23.9 °C, a mean temperature of 18.8 °C, and a minimum temperature of 10.7 °C (Table 4.1.1). The mid-depth sonde had a maximum temperature of 24.8 °C, a mean temperature of 20.3 °C, and a minimum temperature of 10.8 °C. Minimum temperatures at the Brown's Pool station were observed during the barrier beach closure in late November when freshwater displaced the brackish water at the station (Figure 4.1.9). However, temperatures were observed to be lower at the bottom sonde compared to the mid-depth sonde when brackish water was present at the bottom sonde during open conditions (Figure 4.1.18). Under open conditions, warmer freshwater from the MBA would periodically replace the cooler brackish water that was present at the bottom of the pool, resulting in higher temperatures, including the maximum temperature observed on 30 June (Figure 4.1.18). By contrast, temperatures were observed to increase during the closure in October as warm brackish water migrated to the station and displaced the cooler freshwater (Figures 4.1.9 and 4.1.18). Temperatures were then observed to decrease between the subsequent closures as the brackish water was displaced by cooler freshwater.

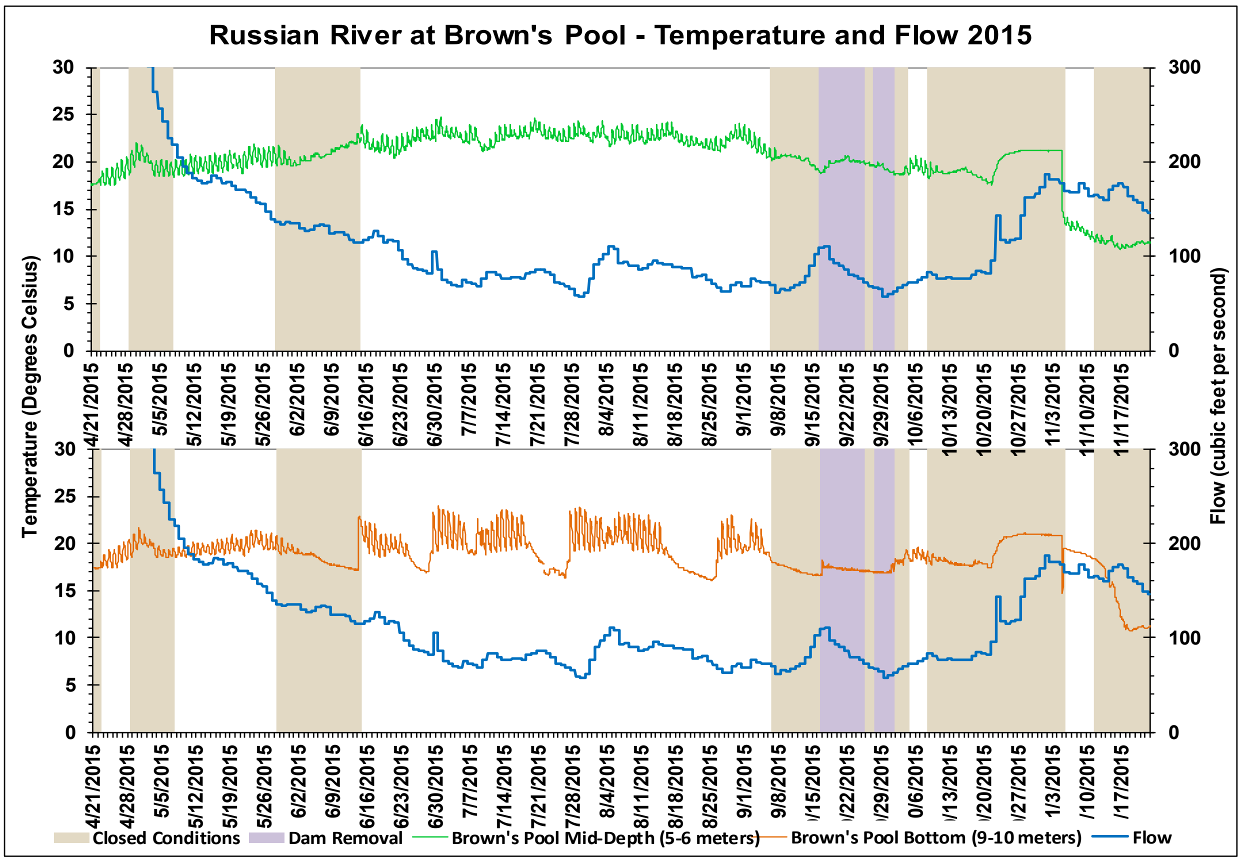


Figure 4.1.18. 2015 Russian River at Brown's Pool Temperature and Flow Graph

Maximum Backwater Area Temperature

Austin Creek had a maximum temperature of 22.3 °C, a mean temperature of 15.8 °C, and a minimum temperature of 7.3 °C (Table 4.1.1). A gradual increase in temperature through the summer months of the Estuary management period coincided with increases in air temperatures (Figure 4.1.19). Closed estuary conditions did not appear to have a significant effect on the temperatures at the Austin Creek station, but were observed to result in reduced daily fluctuations when compared with open conditions. Otherwise, slight increases and decreases in water temperature during closure events typically coincided with increases and decreases in air temperatures (Figure 4.1.19).

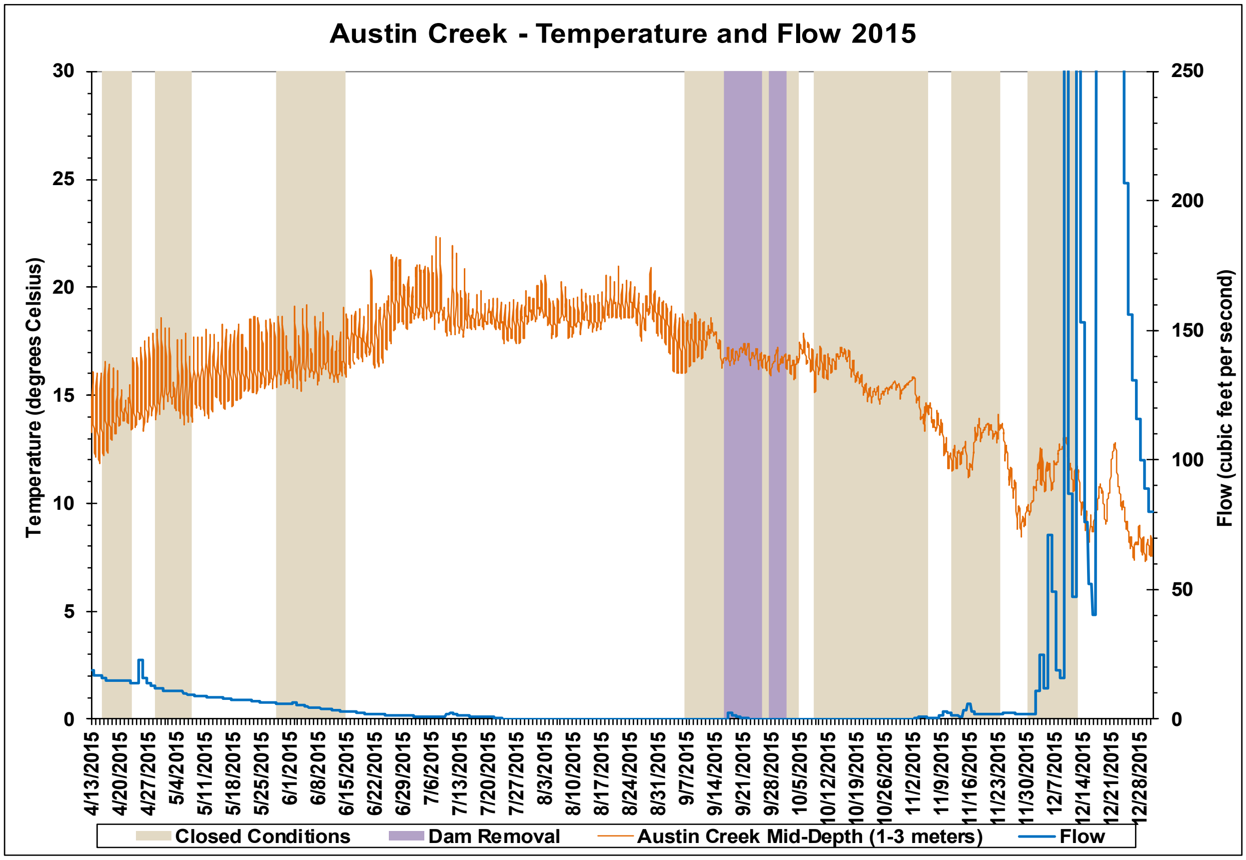


Figure 4.1.19. 2015 Austin Creek Temperature and Flow Graph

Patterson Point had a maximum temperature of 20.7 °C, a mean temperature of 17.4 °C, and a minimum temperature of 10.0 °C (Table 4.1.1). Under open conditions, daily temperatures were lower at Patterson Point than at Brown's Pool in freshwater conditions and at Monte Rio, which suggests that thermal stratification may be occurring at depth (Figure 4.1.19). It is also possible that a groundwater source could be contributing colder water at depth, or it could a combination of both effects occurring in tandem. Daily temperature fluctuations were significantly more stable when compared to Monte Rio (Figure 4.1.21) or Austin Creek before flows became intermittent (Figure 4.1.19), further suggesting some form of thermal stratification or regulation occurring. The station was removed from late July through mid-September due to a lack of consistent access during lower flows. The September barrier beach closure and subsequent closures increased river depths and the station was redeployed through late November. Temperatures continued to decline with atmospheric temperatures through the end of the season and did not appear to be affected by the extended closures (Figure 4.1.20).

The Monte Rio station had a maximum temperature of 27.1 °C, a mean temperature of 20.5 °C, and a minimum temperature of 7.0 °C (Table 4.1.1). Closed Estuary conditions were not observed to have a significant effect on water temperatures at this station, which was consistent with data from previous monitoring efforts at Monte Rio and other monitoring stations within the MBA (Figure 4.1.21). Slight increases and decreases in water temperature during closure events typically coincided with increases and decreases in air temperatures (Figure 4.1.21).

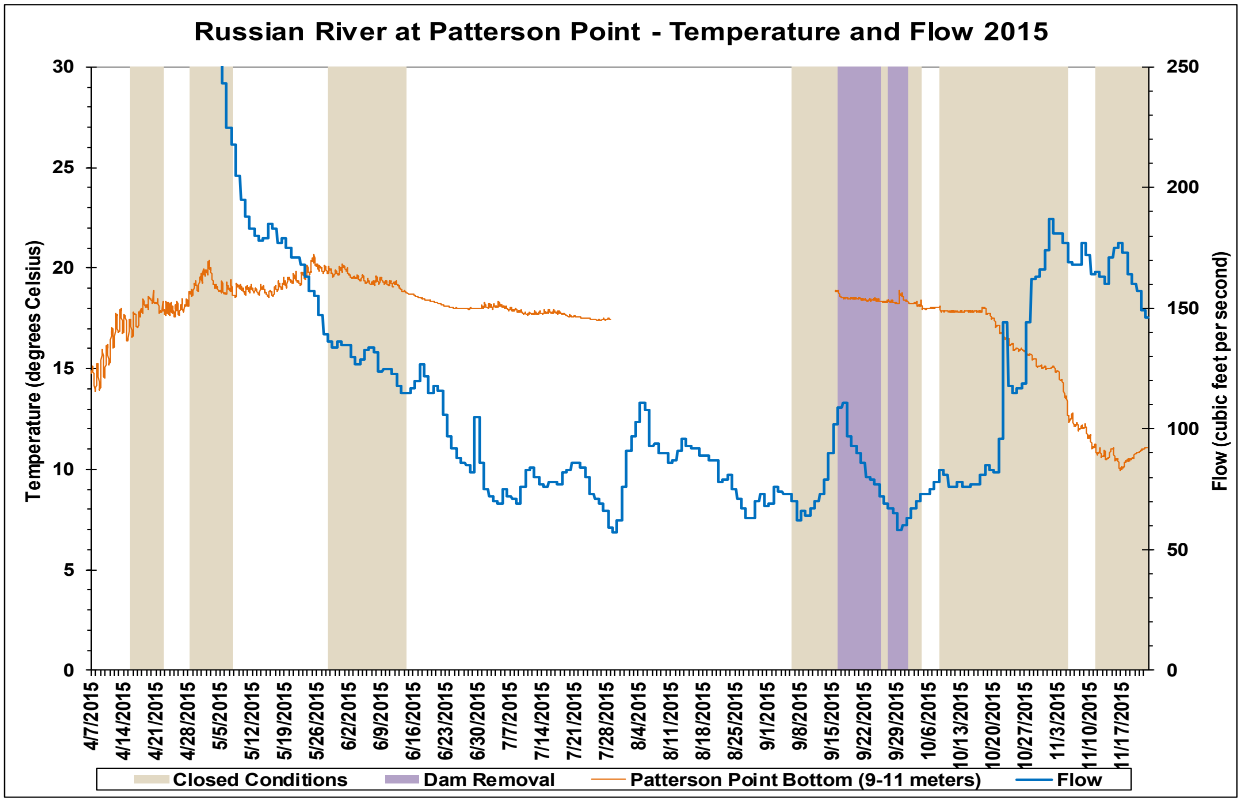


Figure 4.1.20. 2015 Patterson Point Temperature and Flow Graph

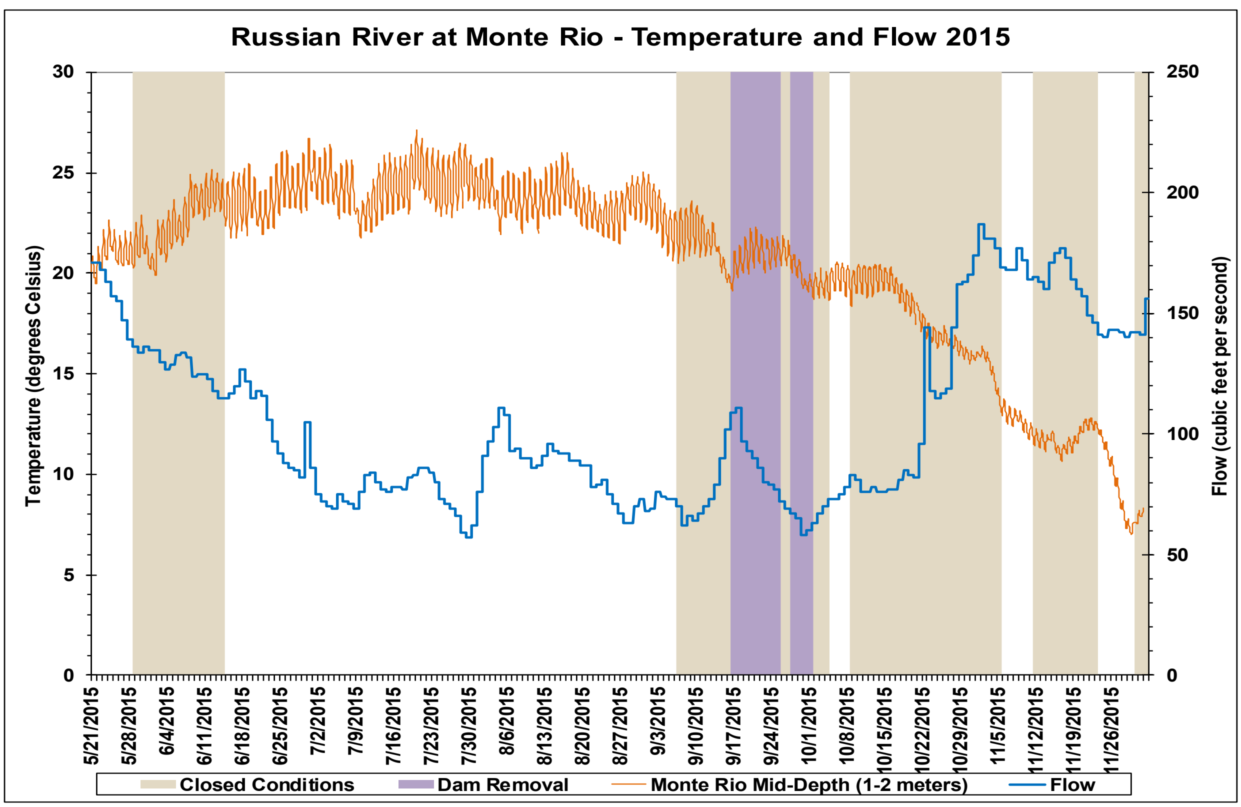


Figure 4.1.21. 2015 Russian River at Monte Rio Temperature and Flow Graph

Dissolved Oxygen

Dissolved oxygen (DO) levels in the Estuary, including the MBA, depend upon factors such as the extent of diffusion from surrounding air and water movement, including freshwater inflow. DO is affected by salinity and temperature stratification, tidal and wind mixing, abundance of aquatic plants, and presence of decomposing organic matter. DO affects fish growth rates, embryonic development, metabolic activity, and under severe conditions, stress and mortality. Cold water has a higher saturation point than warmer water; therefore cold water is capable of carrying higher levels of oxygen.

DO levels are also a function of nutrients, which can accumulate in water and promote plant and algal growth that both consume and produce DO during photosynthesis and respiration. Estuaries tend to be naturally eutrophic because land-derived nutrients are concentrated where runoff enters the marine environment in a confined channel[1]. Upwelling in coastal systems also promotes increased productivity by conveying deep, nutrient-rich waters to the surface, where the nutrients can be assimilated by algae. Excessive nutrient concentrations and plant, algal, and bacterial growth can overwhelm eutrophic systems and lead to a reduction in DO levels that can affect the overall ecological health of the Estuary.

Mean dissolved oxygen concentrations in the lower and middle reaches were generally higher at the surface sondes compared to the mid-depth sondes at a given sampling station (Table 4.1.1). Although the mid-depth and surface sondes were observed to experience supersaturation conditions, the mid-depth sondes also experienced more frequent hypoxic and anoxic conditions that served to decrease the mean seasonal value. These supersaturation and hypoxic events were observed during open and closed conditions (Figures 4.1.22 through 4.1.24).

Dissolved oxygen concentrations in Willow Creek were observed to fluctuate in response to a variety of events including tidal water movement, saline intrusion, and open or closed Estuary conditions. Hypoxic events were observed to occur almost daily in the presence of brackish water during open conditions from mid-June through early September and were frequently preceded or followed by supersaturation conditions as the day progressed through its diurnal cycle (Figure 4.1.25). Whereas, dissolved oxygen concentrations were observed to steadily decline over a period of days after the barrier beach closed in September and again in October. However, dissolved oxygen concentrations were observed to recover between late season closures as oxygenated saline water migrated back to the station (Figure 4.1.25).

Dissolved oxygen concentrations in the upper reach were influenced by the presence or absence of salinity, with lower minimum and mean DO concentrations observed in brackish water and higher minimum and mean concentrations observed in freshwater, especially during closed conditions. In 2015, the Freezeout Creek station was a predominantly freshwater habitat that was subject to elevated salinity levels as the salt wedge migrated up the Estuary during both open and closed conditions (Figure 4.1.8). The elevated salinity levels were predominantly observed at the bottom sonde, though elevated salinity was also seen at the mid-depth sonde

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[1] National Estuarine Eutrophication Assessment by NOAA National Centers for Coastal Ocean Science (NCCOS) and the Integration and Application Network (IAN), 1999.

during open and closed conditions. The Brown's Pool bottom and mid-depth sondes were observed to remain a predominantly freshwater habitat during the 2015 monitoring season under open and closed conditions, with a few exceptions. Salinity was observed to increase during late season closures at the mid-depth and bottom sondes (Figure 4.1.9). Hypoxic and anoxic conditions at both of these sites were observed to occur in brackish and freshwater conditions, though the anoxia was more persistent in brackish conditions, especially during barrier beach closures (Figures 4.1.26 and 4.1.27).

DO concentrations in the upper reach saline layer were also observed to be lower during open and closed conditions than DO concentrations observed in the saline layer in the lower and middle reaches. This effect was more pronounced at the bottom sondes with prolonged periods of hypoxia and anoxia observed to occur in the presence of salinity. This occurs as the saline layer becomes trapped at the bottom of deep holes where there is less circulation, especially further up in the estuary where the influence of the tidal cycle is reduced.

Lower and Middle Reach Dissolved Oxygen

The stations in the lower and middle reaches experienced significant fluctuations in DO concentrations during open and closed Estuary conditions, with supersaturation, hypoxic conditions, and to a lesser degree, anoxic conditions being observed (Figures 4.1.22 through 4.1.24).

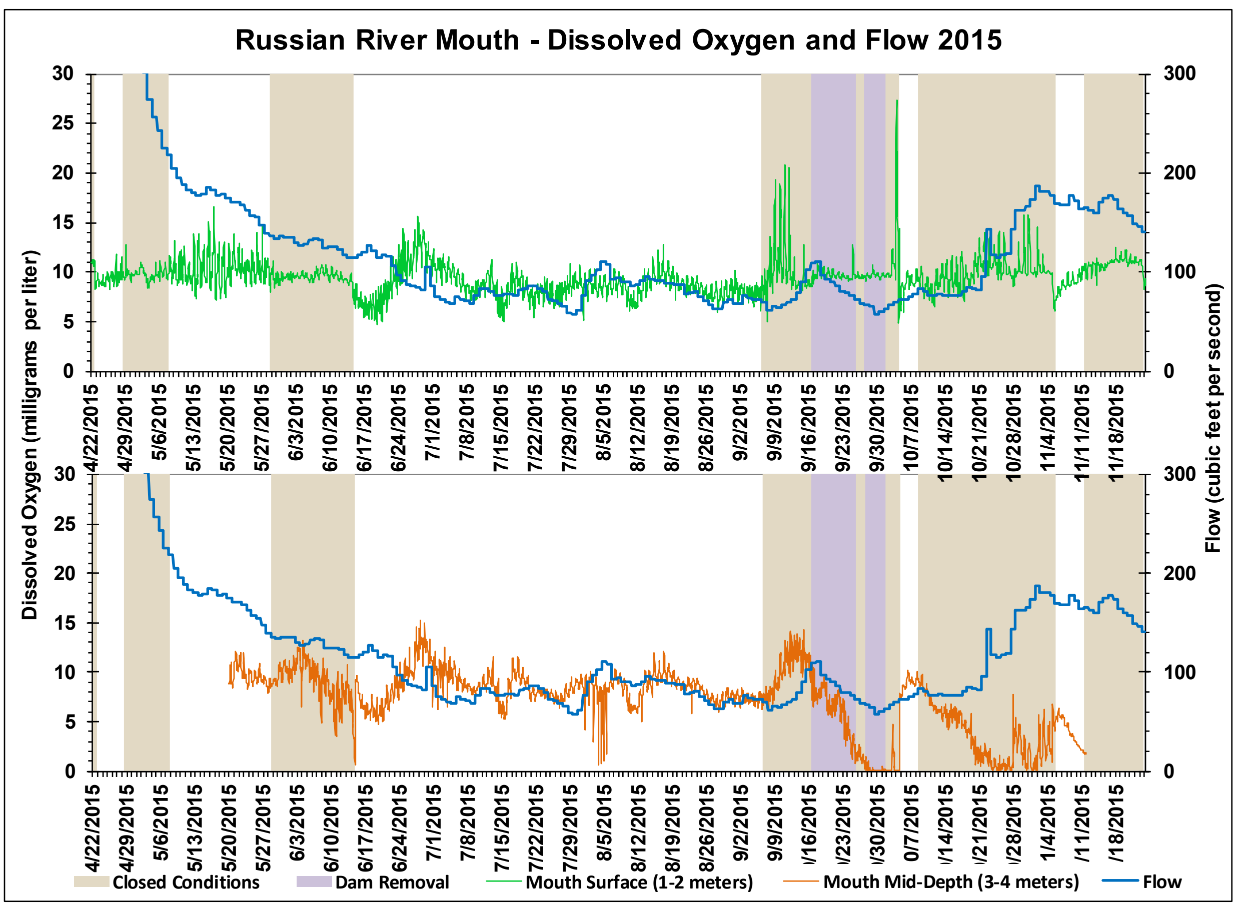


Figure 4.1.22. 2015 Russian River Mouth Dissolved Oxygen and Flow Graph

The surface sondes were observed to have higher mean DO concentrations when compared to the mid-depth sondes (Table 4.1.1). The surface sondes at the Mouth and Patty's Rock had mean DO concentrations of 9.5 and 9.6 mg/L, respectively. Whereas, the mid-depth sondes had mean DO concentrations of 7.2, 8.1, and 7.0 mg/L at the Mouth, Patty's Rock, and Sheephouse Creek stations, respectively (Table 4.1.1).

The effect of closed conditions at the surface sondes was variable as DO concentrations were observed to remain unaffected, slightly decline, or increase in some instances (Figures 4.1.22 and 4.1.23). The Mouth and Patty's Rock surface sondes had minimum DO concentrations of 4.8 and 5.0 mg/L (Table 4.1.1). The minimum concentrations were observed at the Mouth and Patty's Rock surface sondes during open conditions and shortly after the barrier beach reopened in early October (Figures 4.1.22 and 4.1.23).

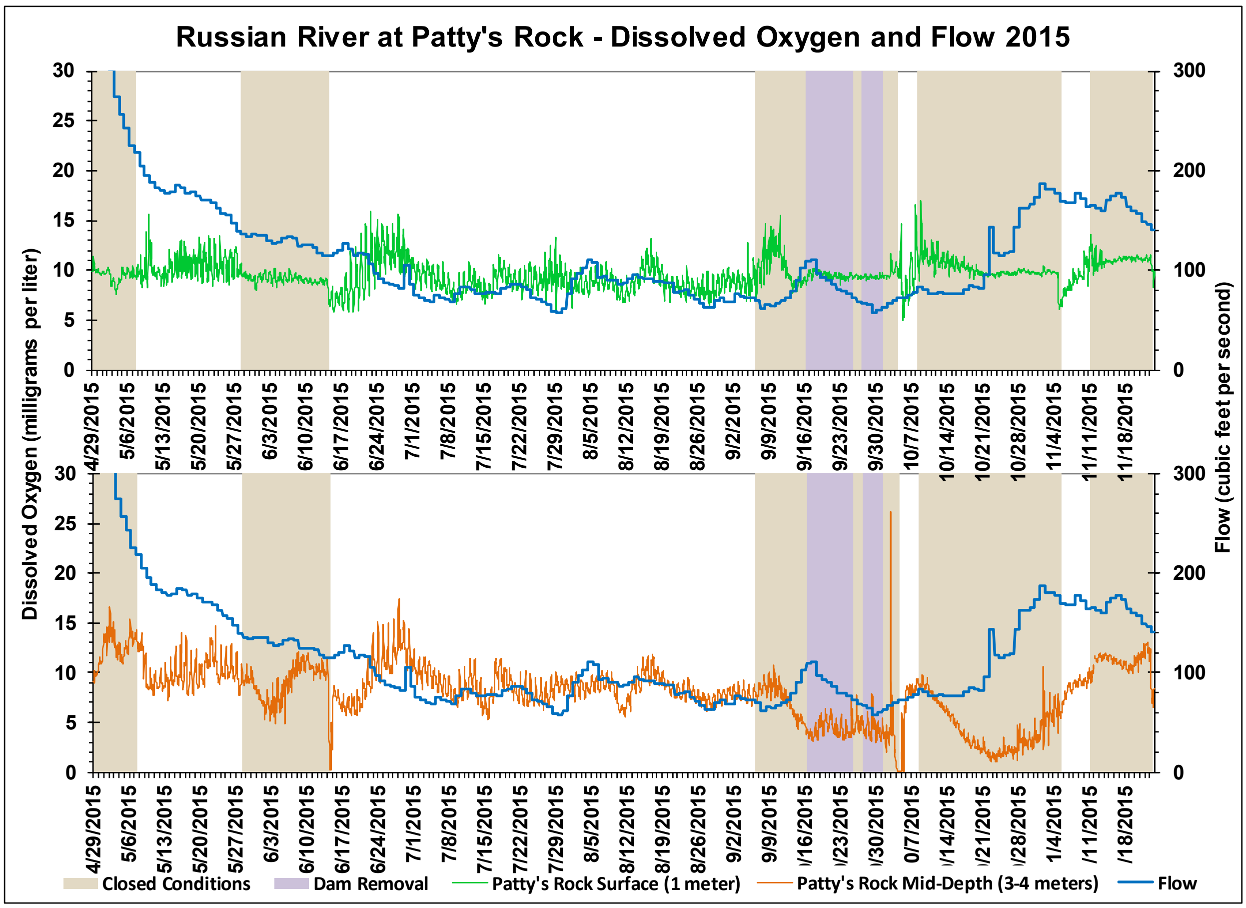


Figure 4.1.23. 2015 Russian River at Patty's Rock Dissolved Oxygen and Flow Graph

DO concentrations were observed to become hypoxic and anoxic at the mid-depth stations during river closures (Figures 4.1.23 and 4.1.24). Corresponding minimum concentrations of DO at the mid-depth sondes were 0.1 mg/L at the Mouth, Patty's Rock, and Sheephouse Creek stations, respectively (Table 4.1.1). As can be seen from these minimum DO concentrations, lower minimum oxygen levels were observed at the mid-depth sondes than at the surface sondes.

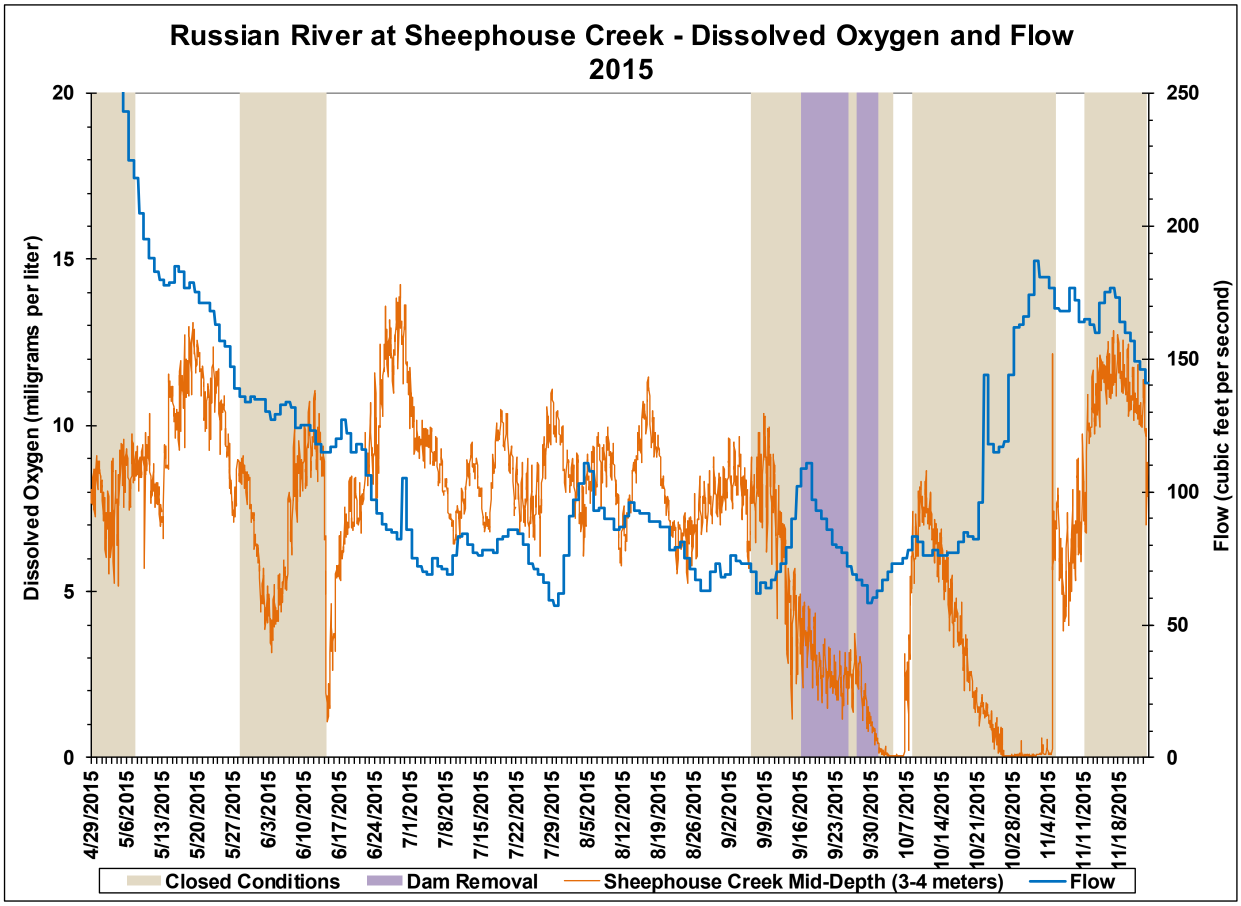


Figure 4.1.24. 2015 Russian River at Sheephouse Creek Dissolved Oxygen and Flow Graph

The lower and middle reach surface sondes, and mid-depth sondes to a lesser degree, experienced hourly fluctuating supersaturation events. Supersaturation events were observed at the surface and mid-depth sondes during open and closed estuary conditions (Figures 4.1.22 through 4.1.24). At times when oxygen production exceeds the diffusion of oxygen out of the system, supersaturation may occur (Horne, 1994). DO concentrations exceeding 100% saturation in the water column are considered supersaturated conditions. Because the ability of water to hold oxygen changes with temperature, there are a range of concentration values that correspond to 100% saturation. For instance, at sea level, 100% saturation is equivalent to approximately 11 mg/L at 10 °C, but only 8.2 mg/L at 24 °C. Consequently, these two temperature values roughly represent the range of temperatures typically observed in the Estuary.

The Mouth surface sonde had a maximum DO concentration of 27.4 mg/L, which corresponded to 316% saturation. The maximum DO concentration at the Patty's Rock surface sonde was 17.0 mg/L, or 212% saturation (Table 4.1.1). Maximum DO concentrations at the mid-depth sondes were approximately 15.2 mg/L (177%) at the Mouth, 26.1 mg/L (318%) at Patty's Rock, and 14.2 mg/L (172%) at Sheephouse Creek, respectively (Table 4.1.1).

The Willow Creek sonde had a minimum DO concentration of 0.0 mg/L, a mean DO concentration of 6.5 mg/L, and a maximum DO concentration of 18.8 mg/L (238%) (Table 4.1.1). Frequent fluctuations between hypoxic and supersaturated DO concentrations were observed during open conditions after brackish water migrated into Willow Creek in May (Figure 4.1.25). Hypoxic and anoxic conditions were also observed to occur in brackish water during Estuary closures. (Figure 4.1.25).

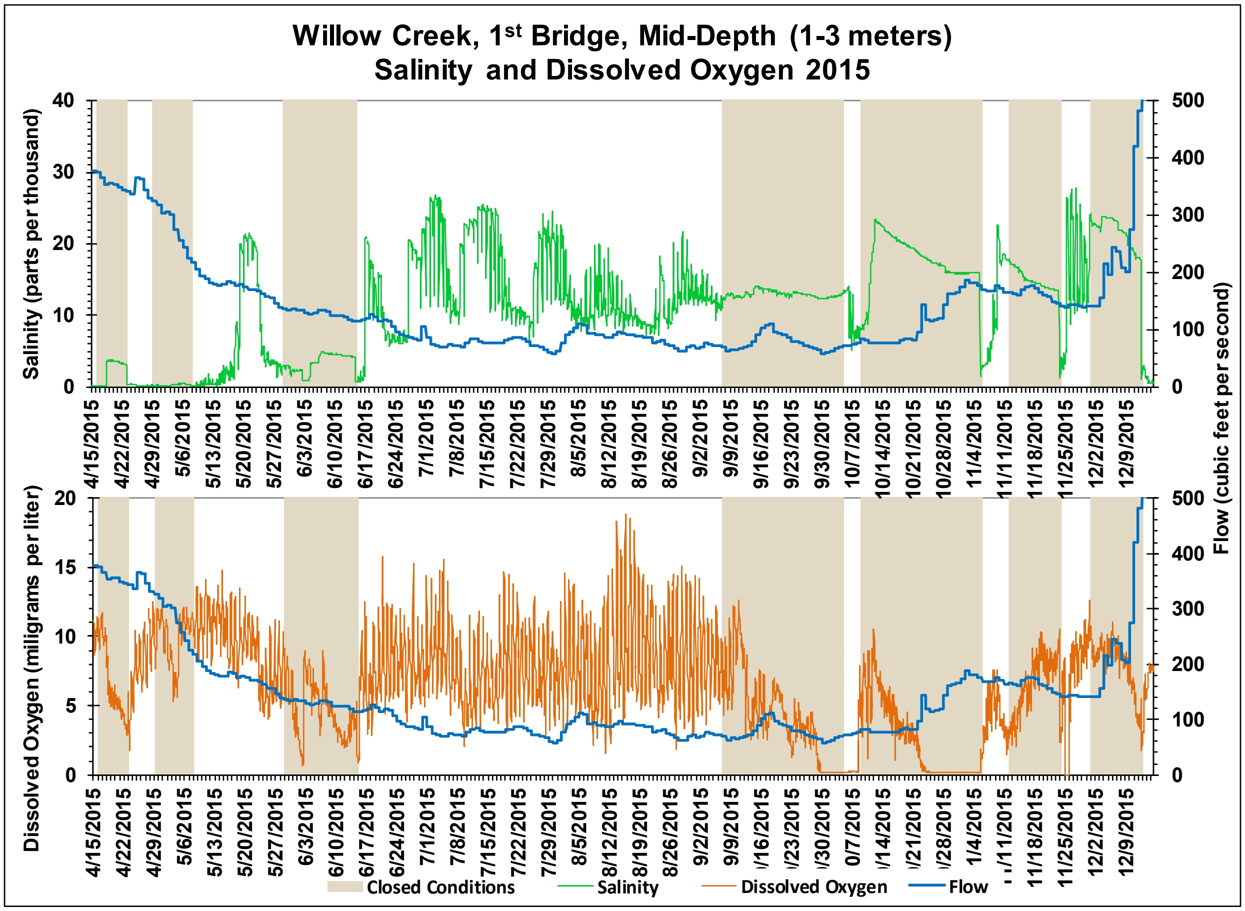


Figure 4.1.25. 2015 Willow Creek Salinity and Dissolved Oxygen Graph

Upper Reach Dissolved Oxygen

The Freezeout Creek bottom sonde had a minimum concentration of 0.0 mg/L, a mean DO concentrations of 5.0 mg/L, and a maximum concentration of 13.5 mg/L (155%) (Table 4.1.1). The mid-depth sonde at Freezeout Creek had a minimum concentration of 0.1 mg/L, a mean DO concentration of 7.1 mg/L, and a maximum concentration of 16.9 mg/L (215%) (Table 4.1.1).

DO concentrations at the Freezeout Creek bottom sonde fluctuated significantly and became hypoxic and anoxic during open and closed Estuary conditions when saline water was present (Figure 4.1.26). The bottom was predominantly freshwater during open and closed conditions through mid-June with minor fluctuations in salinity concentrations of less than 1 ppt. The Freezeout Creek bottom sonde then transitioned to a primarily brackish habitat from mid-June through August, until an equipment malfunction occurred and the sonde was removed for service. These fluctuations in salinity concentration often occurred on a daily and even hourly basis. DO typically fluctuated with changing salinity concentrations, becoming depressed in saline water and recovering in freshwater (Figure 4.1.26). The Freezeout Creek bottom sonde was redeployed in late October during a closed Estuary with brackish water and anoxic conditions. DO concentrations briefly recovered after the Estuary reopened in early November as freshwater briefly replaced the brackish water at the station (Figure 4.1.26). However, DO concentrations declined as brackish water returned to the station and remained anoxic through the subsequent closure (Figure 4.1.26).

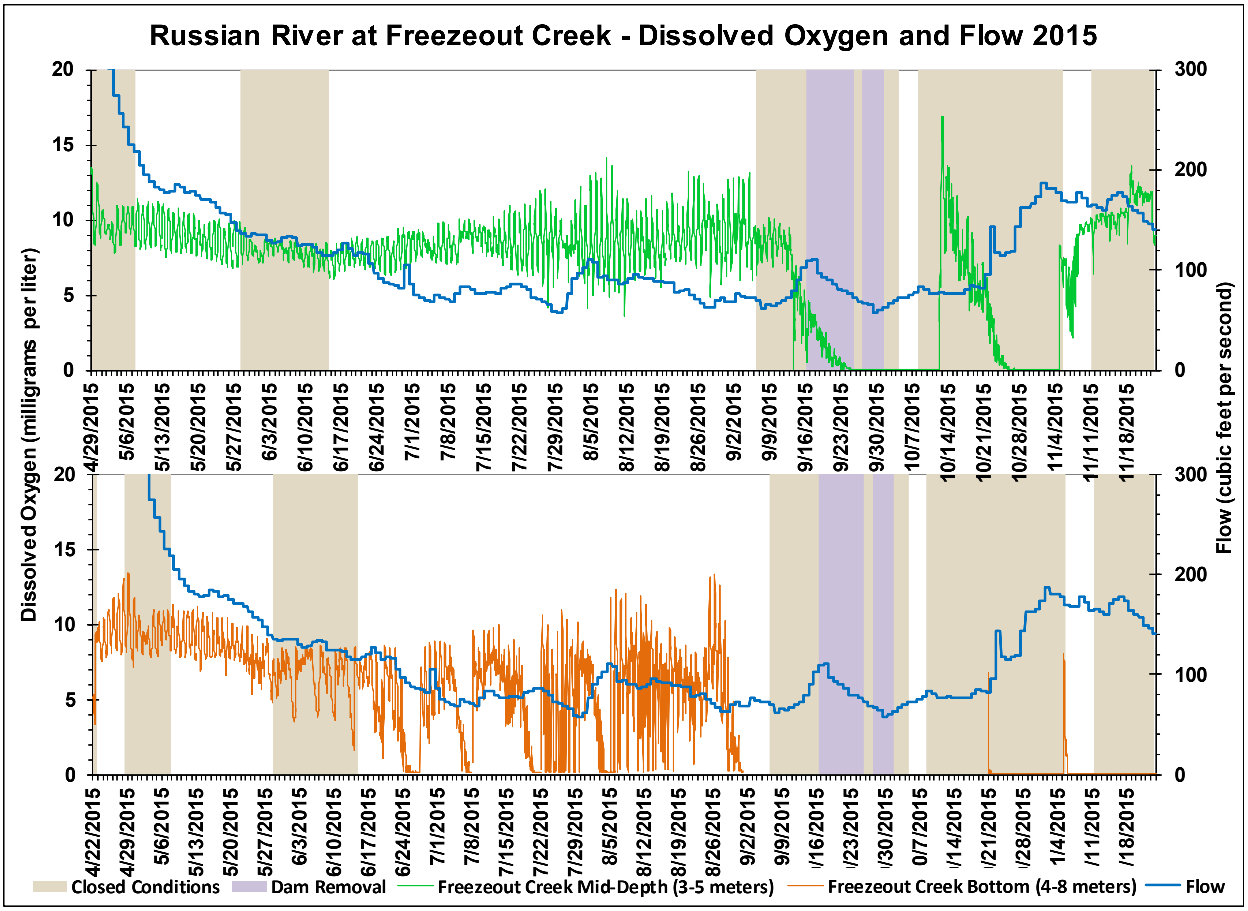


Figure 4.1.26. 2015 Russian River at Freezeout Creek Dissolved Oxygen and Flow Graph

The Freezeout Creek mid-depth sonde was also observed to have brackish conditions during open conditions from mid-June through early September, though to a far lesser degree than the bottom sonde (Figure 4.1.8). These brackish conditions were below 5 ppt, which is less than the bottom sonde, and occurred less frequently. DO concentrations were observed to remain stable at the mid-depth sonde in freshwater conditions, but became anoxic and hypoxic in the presence of brackish water during and between Estuary closures from September through early November (Figure 4.1.26). Conversely, DO concentrations recovered after the October closure as freshwater replaced the brackish water at the mid-depth sonde. DO concentrations then became supersaturated at the mid-depth sonde during the November closure as an oxygenated layer of salt water migrated into the mid-depth of the water column (Figure 4.1.26).

The Brown's Pool bottom sonde had a minimum concentration of 0.1 mg/L, a mean DO concentration of 2.8 mg/L, and a maximum concentration of 10.2 mg/L (116%) (Table 4.1.1). The Brown's Pool mid-depth sonde had a minimum concentration of 0.1 mg/L, a mean DO concentration of 7.9 mg/L, and a maximum concentration of 12.2 mg/L (139%) (Table 4.1.1). The Brown's Pool bottom and mid-depth sondes were observed to remain a predominantly freshwater habitat during the 2015 monitoring season under open and closed conditions, with a few exceptions. The bottom and mid-depth of Brown's Pool was predominantly freshwater during the entire monitoring season in open and closed conditions (Figure 4.1.9). However, there were frequent brief periods of brackish conditions observed at the bottom sonde during open and closed conditions. As such, DO concentrations at the bottom sonde were observed to fluctuate between anoxic and normal concentrations. As saline water migrated into the station during the October closure, oxygen levels were observed to decline at the mid-depth and bottom sondes. However, once the freshwater conditions returned to the mid-depth, oxygen levels were observed to recover. (Figures 4.1.9 and 4.1.27).

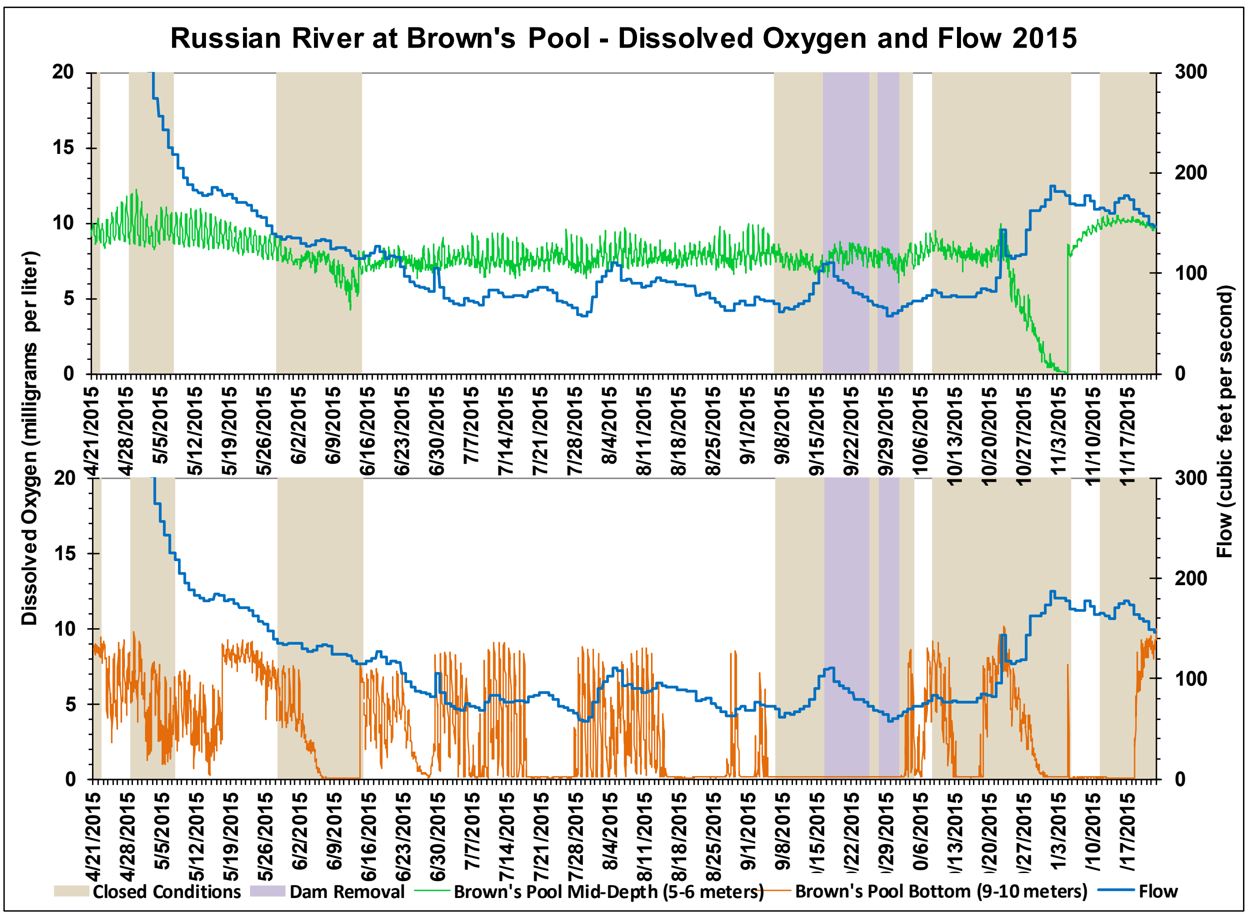


Figure 4.1.27. 2015 Russian River at Brown's Pool Dissolved Oxygen and Flow Graph

Maximum Backwater Area Dissolved Oxygen

The Austin Creek station had minimum, mean, and maximum DO concentrations of 0.1, 5.0, and 11.1 (115%) mg/L, respectively (Table 4.1.1). Similar to previous monitoring seasons, DO concentrations in 2015 gradually declined through the summer months as flows decreased and mixing was significantly reduced (Figure 4.1.28). As a result of continuing drought conditions, flows became intermittent earlier in 2015 than in 2013 or 2014 measuring less than 2 cfs at the upstream USGS gauging station by late June. The sonde was now in an isolated pool where DO concentrations became hypoxic. Minimum values at Austin Creek were observed during open conditions in July and during an Estuary closure in September (Figure 4.1.28). Interestingly, as the closed estuary filled and began to inundate the Austin Creek station, DO concentrations showed signs of recovery, with daily fluctuations from anoxic to slightly hypoxic conditions increasing over time to a maximum of approximately 10 mg/L by the end of November. However, DO concentrations did not begin to fully recover to springtime levels until storm related flows began to increase in December (Figure 4.1.28). Summer dam removal did not appear to have a negative effect on DO concentrations. The station was hypoxic to anoxic before removal began on 17 September and conditions actually began to improve during and following dam removal.

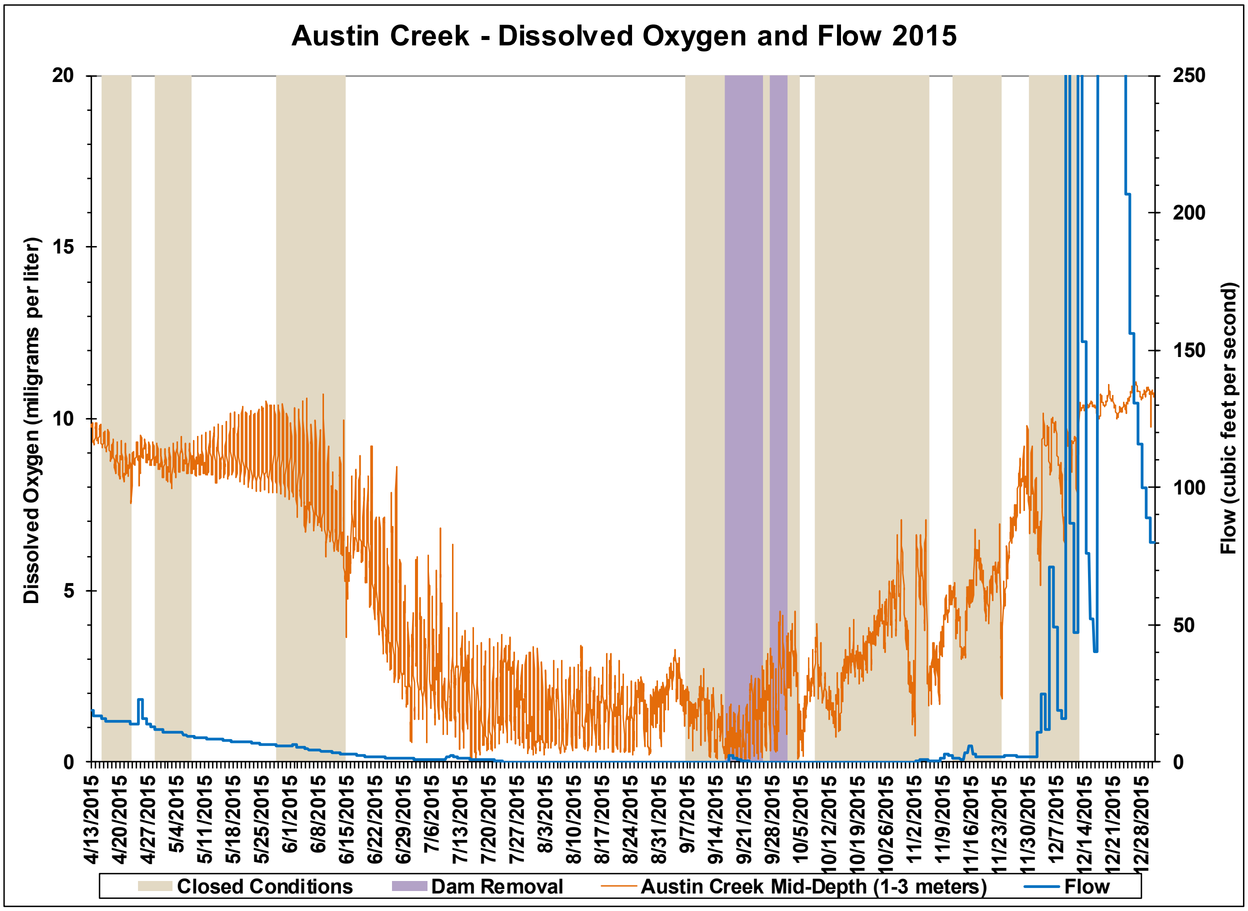


Figure 4.1.28. 2015 Austin Creek Dissolved Oxygen and Flow Graph

DO response to estuary closures was variable at the Austin Creek station. Concentrations were observed to initially decline during the closure in September, but were also observed to increase during the same closure and following summer dam removal. Concentrations began to decline again in mid-October and became variable as the barrier beach was breached and then closed again. However concentrations were higher during the closures than during open conditions when flows were intermittent (Figure 4.1.28).

Patterson Point had a minimum concentration of 0.1 mg/L, a mean concentration of 4.1 mg/L, and a maximum concentration of 10.6 (109%). The station is located at the bottom of a deep pool and remained predominantly hypoxic to anoxic throughout the monitoring season under both open and closed conditions. Frequent fluctuations in DO concentrations were observed during higher spring flows, but the station became anoxic during the June closure and remained anoxic during open conditions until the sonde was removed in late July. Concentrations were observed to recover during closed conditions from mid-October through late November as storm flows increased (Figure 4.1.29).

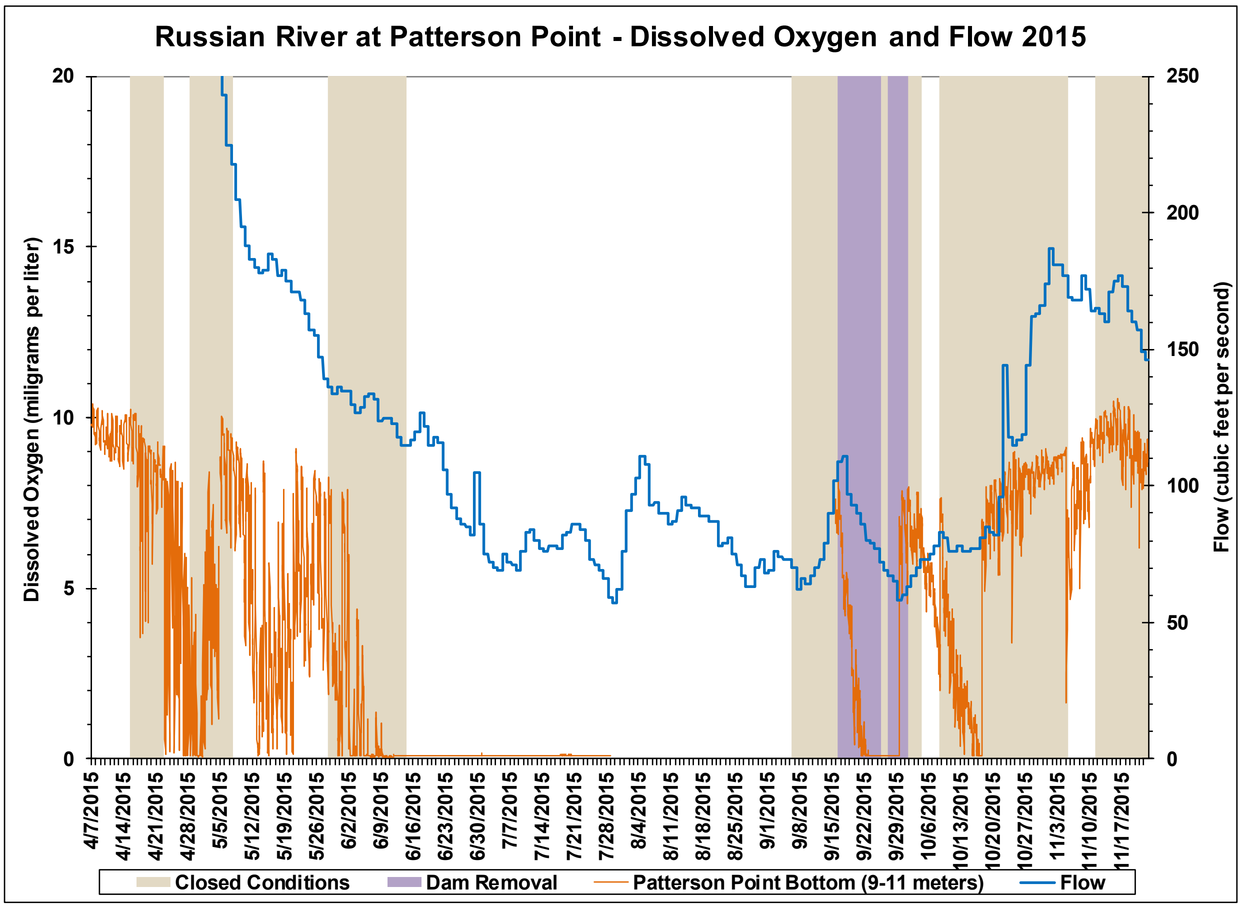


Figure 4.1.29. 2015 Russian River at Patterson Point Dissolved Oxygen and Flow Graph

The Monte Rio Station had a minimum concentration of 6.2 mg/L, a mean DO concentration of 8.6 mg/L, and a maximum concentration of 12.1 mg/L (118%) (Table 4.1.1). The minimum DO concentration occurred on 10 July during open conditions (Figure 4.1.30). Although there were some temporally localized DO concentrations between 6 and 8 mg/L, DO concentrations did not appear to be significantly affected by summer flows or closed conditions and remained above 8 mg/L, on average, during both open and closed conditions (Figure 4.1.30).

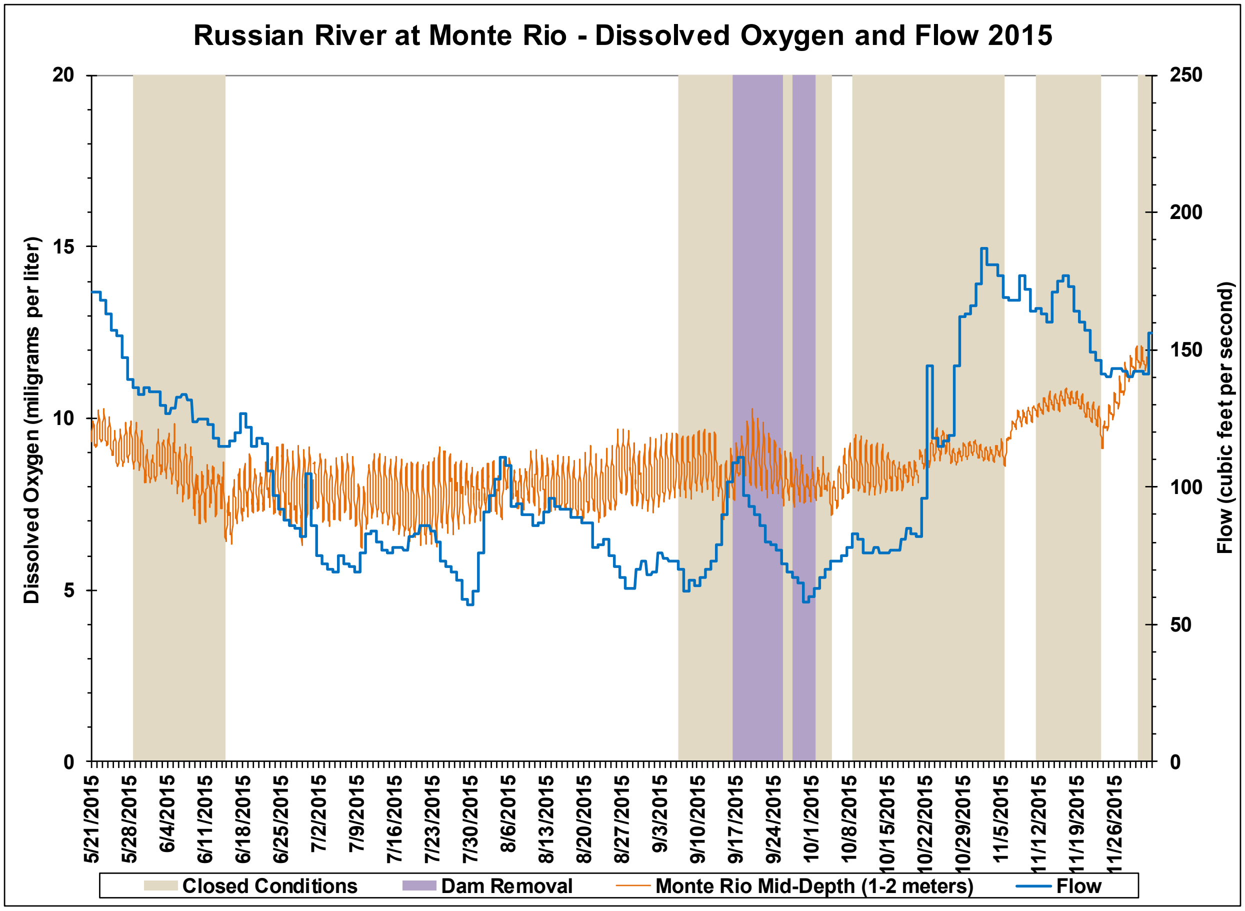


Figure 4.1.30. 2015 Russian River at Monte Rio Dissolved Oxygen and Flow Graph

Hydrogen Ion (pH)

The acidity or alkalinity of water is measured in units called pH, an exponential scale of 1 to 14 (Horne, 1994). Acidity is controlled by the hydrogen ion H+, and pH is defined as the negative log of the hydrogen ion concentration. A pH value of 7 is considered neutral, freshwater streams generally remain at a pH between 6 and 9, and ocean derived salt water is usually at a pH between 8 and 9. When the pH falls below 6 over the long term, there is a noticeable reduction in the abundance of many species, including snails, amphibians, crustacean zooplankton, and fish such as salmon and some trout species (Horne 1994).

Lower and Middle Reach pH

Mean hydrogen ion (pH) values were fairly consistent among all mid-depth stations in the lower and middle reaches, with values of 7.8, 8.1, and 7.8 pH observed at the Mouth, Patty's Rock, and Sheephouse Creek, respectively (Figures 4.1.31 through 4.1.33). The Mouth and Patty's Rock surface sondes were also consistent, with mean pH values of 8.3 and 8.0 pH, respectively (Table 4.1.1).

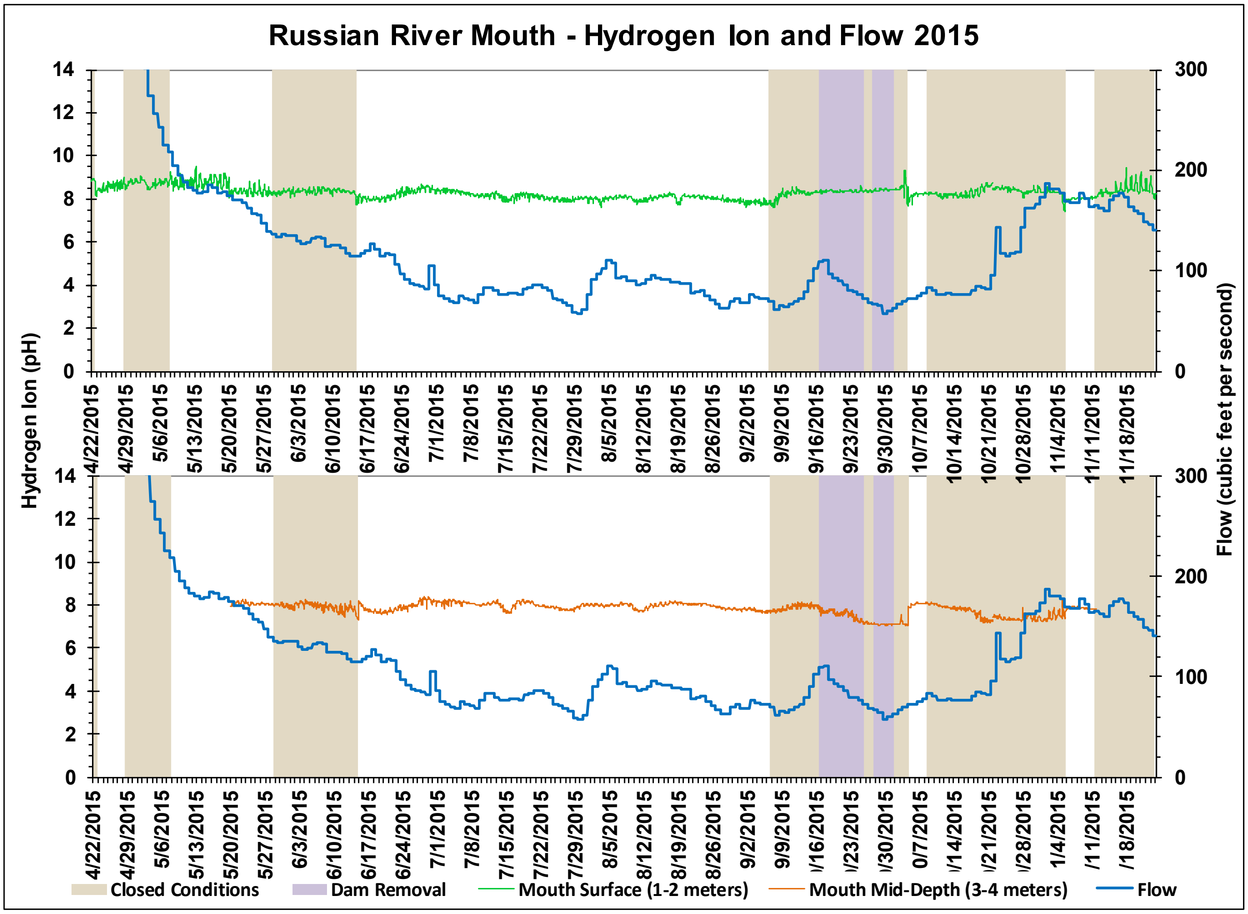


Figure 4.1.31. 2015 Russian River Mouth Hydrogen Ion and Flow Graph

Maximum and minimum pH values were also fairly consistent across stations in the lower and middle reaches at both mid-depth and at the surface. Maximum pH values at the Mouth, Patty' Rock, and Sheephouse Creek mid-depth sondes were observed to be 8.4, 9.0, and 8.5 pH, respectively. Maximum pH values at the Mouth and Patty's Rock surface sondes were observed to be 9.5 and 9.4 pH, respectively. Minimum pH values at the mid-depth sondes were 7.1, 7.1, and 7.0 pH at the Mouth, Patty's Rock, and Sheephouse Creek, respectively. Similarly, the minimum pH values at the surface sondes were observed to be 7.4 and 7.2 pH at the Mouth and Patty's Rock, respectively.

Although minimum, mean, and maximum pH values were fairly consistent amongst the lower and middle reach stations, pH values were observed to vary with increases and decreases of DO concentrations, with higher values generally observed during supersaturation conditions and lower values during hypoxic conditions (Figures 4.1.31 through 4.1.33). This was especially apparent when pH values were as high as 9.5 at the Mouth surface sonde during a supersaturation event in May when the estuary was open (Figure 4.1.31).

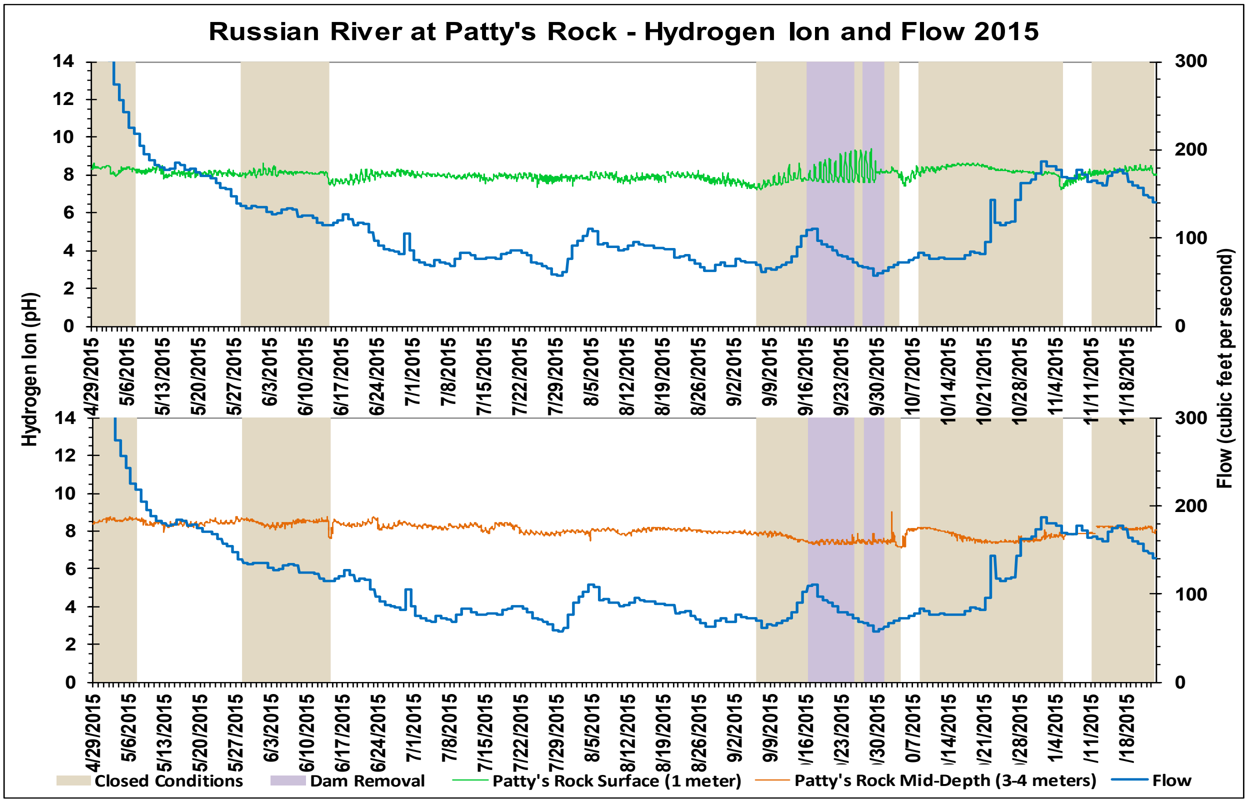


Figure 4.1.32. 2015 Russian River at Patty's Rock Hydrogen Ion and Flow Graph

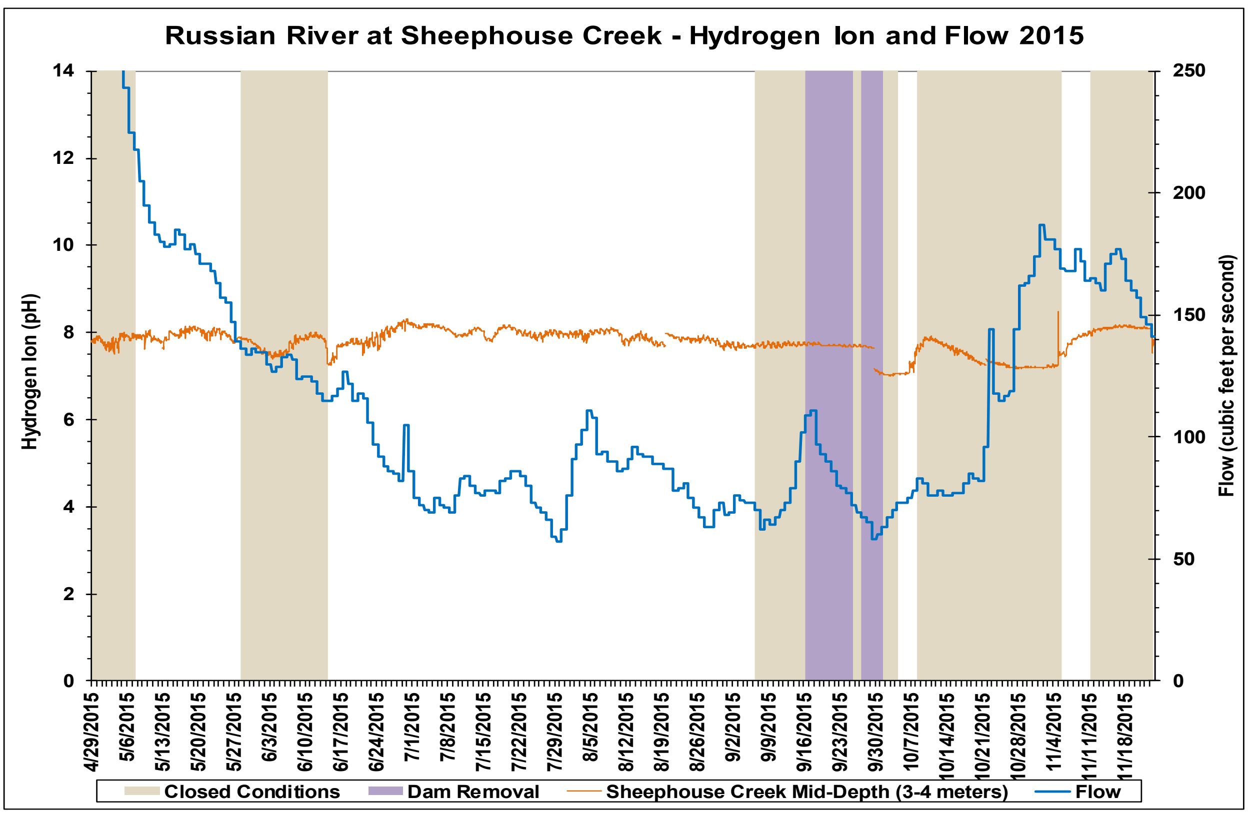


Figure 4.1.33. 2015 Russian River at Sheephouse Creek Hydrogen Ion and Flow Graph

The Willow Creek station had a minimum pH value of 6.4, a mean pH value of 7.6, and a maximum pH value of 8.8 (Table 4.1.1). The Willow Creek station also had pH values that were observed to vary with increases and decreases of DO concentrations, as well as with fluctuations in salinity associated with reduced freshwater flows, tidal influence, and Estuary closures (Figures 4.1.25 and 4.1.34). Minimum pH values were observed to occur after the estuary reopened in June and following the late season closures. Maximum values were observed in mid-summer during open conditions in brackish water.

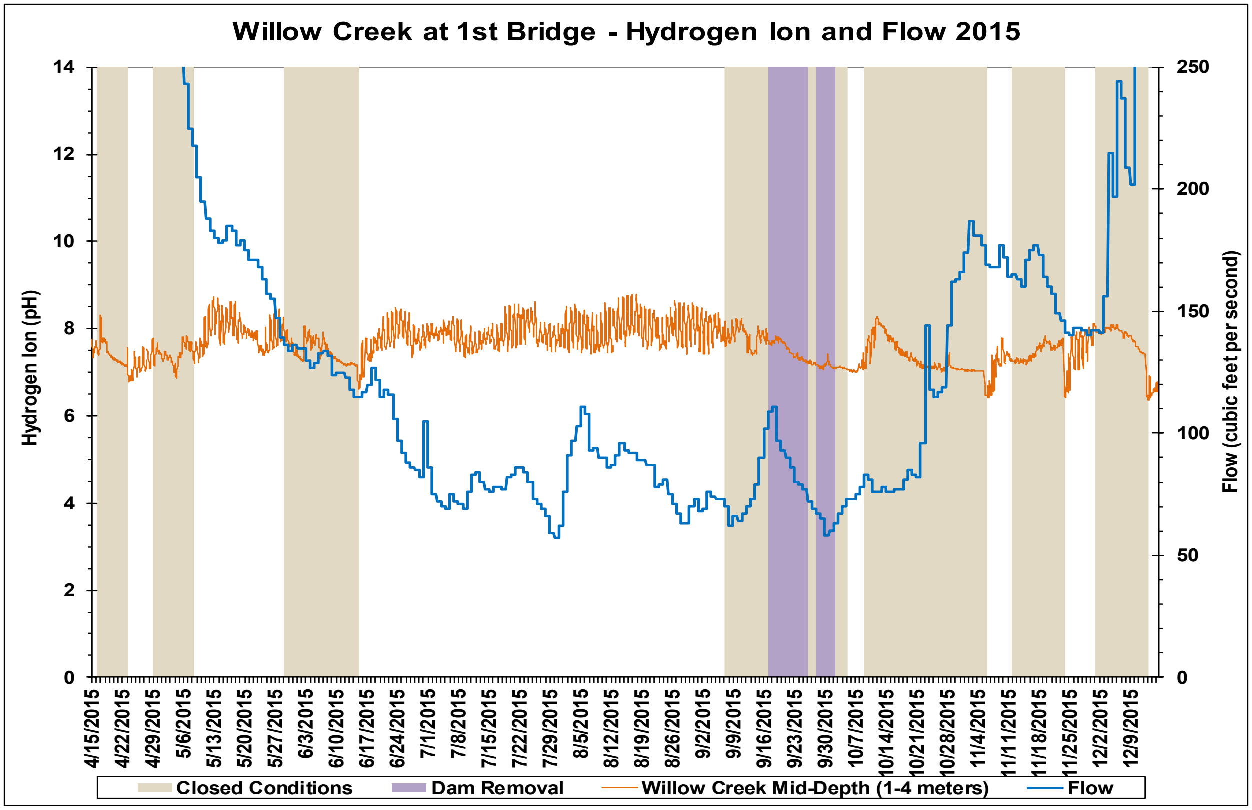


Figure 4.1.34. 2015 Willow Creek Hydrogen Ion and Flow Graph

Upper Reach pH

The Freezeout Creek bottom sonde recorded a minimum pH value of 6.4, a mean pH value of 7.5, and a maximum pH value of 8.9 (Table 4.1.1). The Freezeout Creek mid-depth sonde recorded a minimum pH value of 7.0, a mean pH value of 7.9, and a maximum pH value of 8.9 (Table 4.1.1). The Freezeout Creek station had pH values that were observed to vary with DO concentrations in the presence of both freshwater and brackish water (Figures 4.1.26 and 4.1.35).

The Brown’s Pool bottom sonde had a minimum pH value of 6.2, a mean pH value of 7.2, and a maximum pH value of 8.3 (Table 4.1.1). The Brown’s Pool mid-depth sonde had a minimum pH value of 7.1, a mean pH value of 8.0, and a maximum pH value of 8.9 (Table 4.1.1). Minimum pH values occurred at the mid-depth sonde during anoxic conditions when the Estuary was closed (Figures 4.1.27 and 4.1.36). Whereas, minimum pH values occurred at the bottom sonde during anoxic conditions then the Estuary was open (Figures 4.1.26 and 4.1.36).

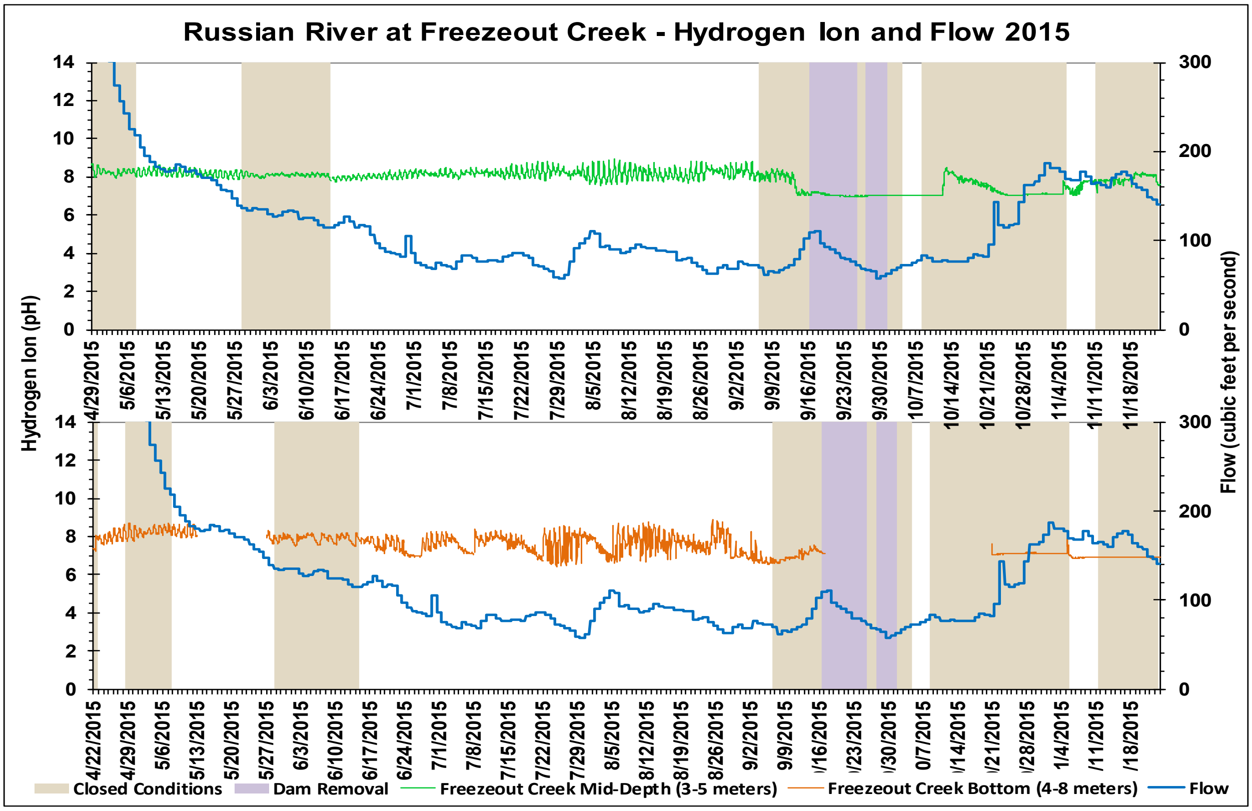


Figure 4.1.35. 2015 Russian River at Freezeout Creek Hydrogen Ion and Flow Graph

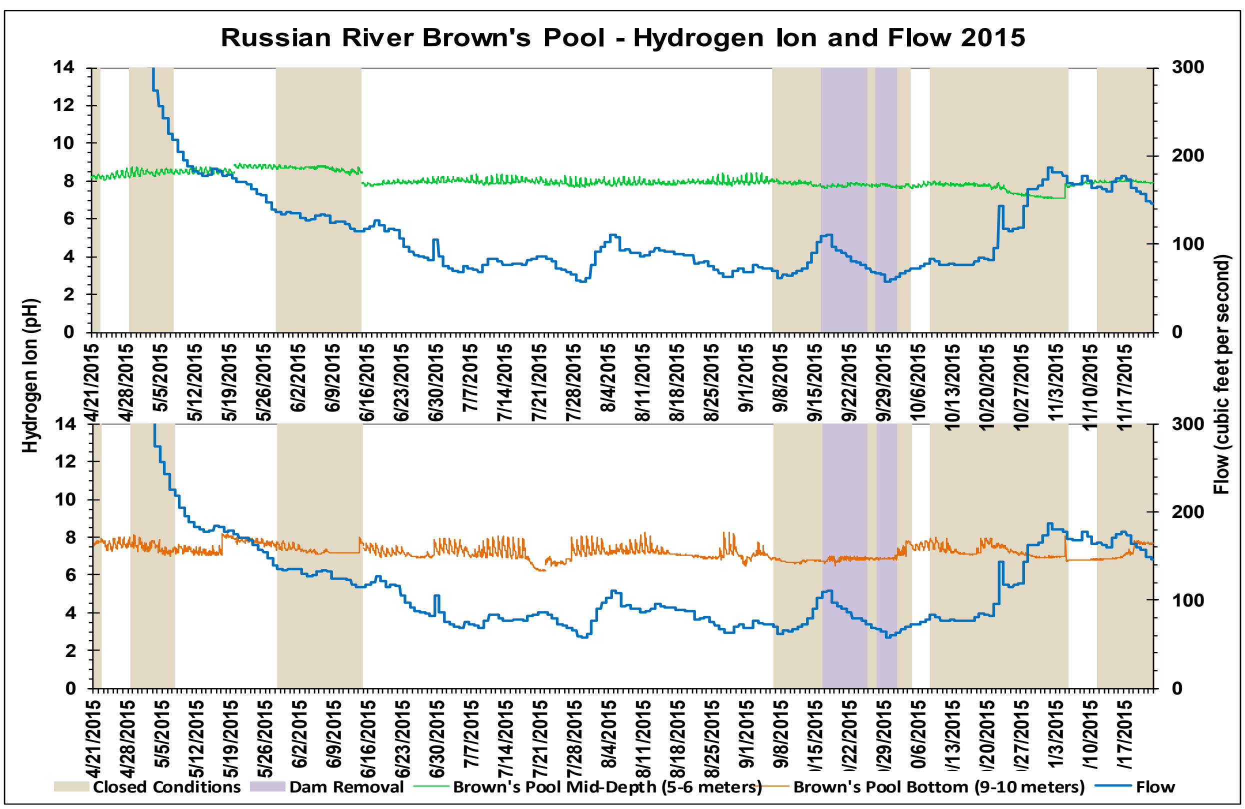


Figure 4.1.36. 2015 Russian River at Brown's Pool Hydrogen Ion and Flow Graph

Maximum Backwater Area pH

The Austin Creek sonde had a minimum pH value of 6.9, a mean pH value of 7.4, and a maximum pH value of 8.0 (Table 4.1.1). The Austin Creek sonde also had pH values that were generally observed to vary with increases and decreases of DO concentrations (Figures 4.1.28 and 4.1.37). Minimum pH values were observed during open and closed Estuary conditions while DO levels were depressed (Figure 4.1.37). Maximum pH values were observed during open and closed Estuary conditions when flows and DO concentrations were higher (Figures 4.1.28 and 4.1.37).

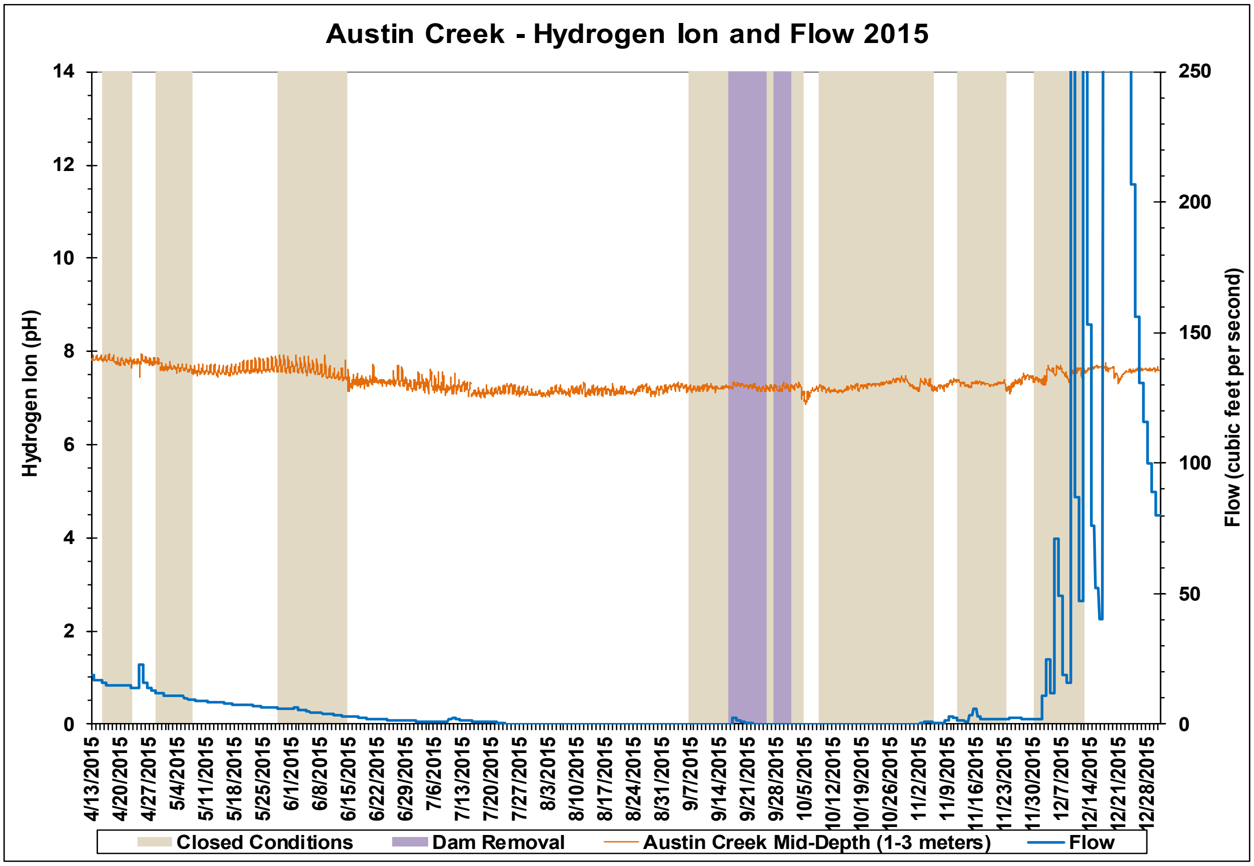


Figure 4.1.37. 2015 Austin Creek Hydrogen Ion and Flow Graph

The Patterson Point sonde had a minimum pH value of 6.3, a mean pH value of 7.2, and a maximum pH value of 8.2 (Table 4.1.1). The Patterson Point sonde also had pH values that were generally observed to vary with increases and decreases of DO concentrations (Figures 4.1.29 and 4.1.38). Minimum concentrations were observed during anoxic conditions when the Estuary was open.

The Monte Rio sonde recorded a minimum pH value of 7.5, a mean pH value of 7.9, and a maximum pH value of 8.5 (Table 4.1.1). Again, the sonde here recorded pH values that were generally observed to vary with increases and decreases of DO concentrations (Figures 4.1.30 and 4.1.39). Overall, pH concentrations did not appear to be significantly affected by summer flows or closed conditions and remained fairly stable through the monitoring period (Figure 4.1.39).

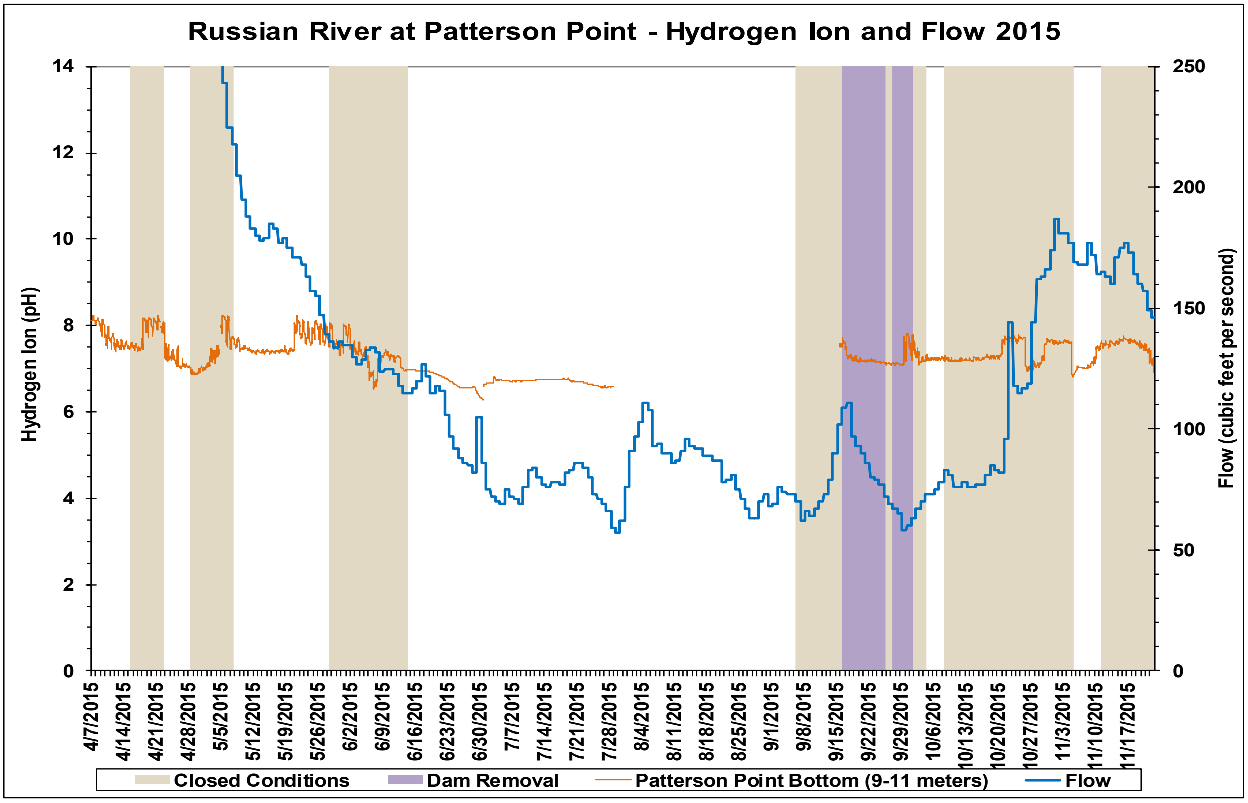


Figure 4.1.38. 2015 Patterson Point Hydrogen Ion and Flow Graph

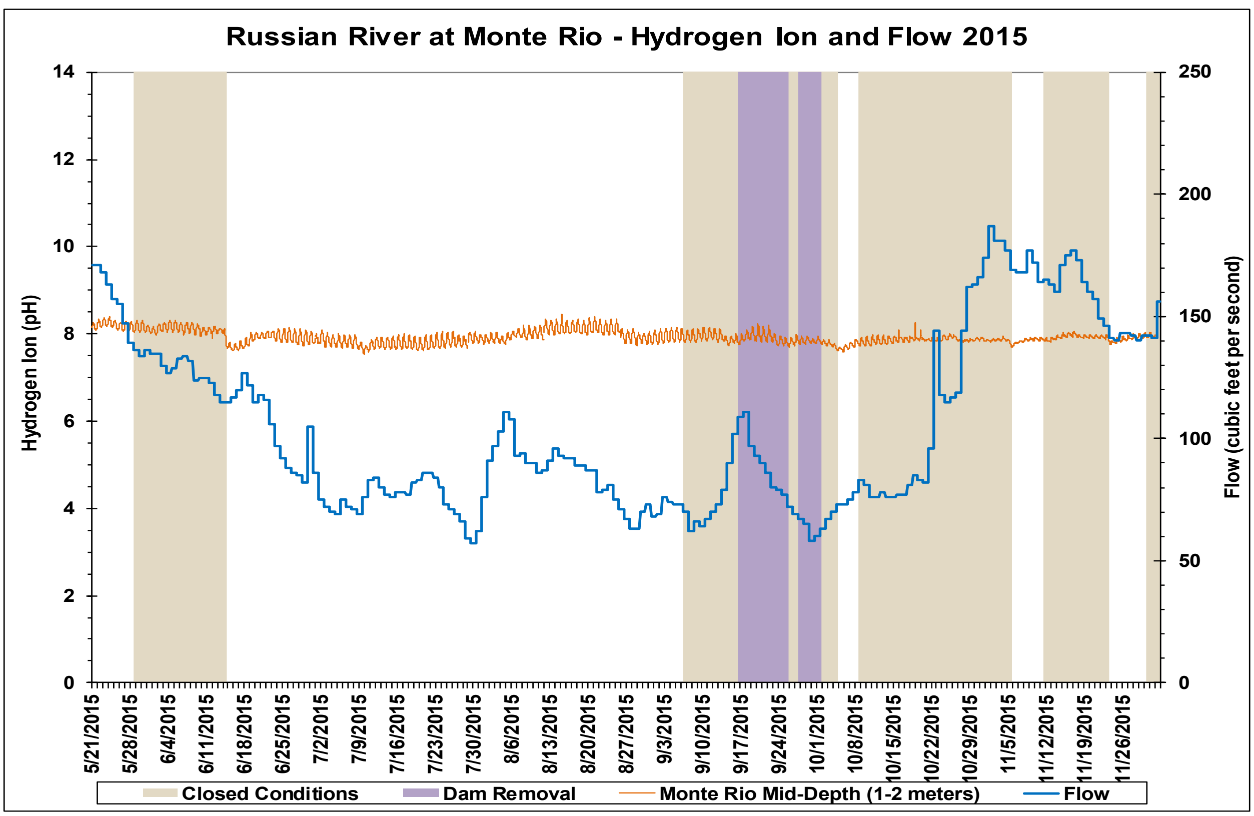


Figure 4.1.39. 2015 Russian River at Monte Rio Hydrogen Ion and Flow Graph

Grab Sampling

Water Agency staff conducted weekly grab sampling from May 12 to October 13 at five stations in the mainstem of the lower river including: Jenner; Casini Ranch; Patterson Point, Monte Rio, and Vacation Beach (Figure 4.1.1). Additional focused sampling was conducted during or after Estuary closures, as well as during summer dam removal in late September, where Agency staff would collect three samples in ten days (Tables 4.1.2 through 4.1.6). Samples collected and analyzed for nutrients, turbidity, chlorophyll a, and indicator bacteria are discussed below. Other sample results including organic carbon, and dissolved solids are not discussed, but are included in Appendix 4.5.

Nutrients

The United States Environmental Protection Agency (USEPA) has established section 304(a) nutrient criteria across 14 major ecoregions of the United States. The Russian River was designated in Aggregate Ecoregion III (USEPA 2013a). USEPA's section 304(a) criteria are intended to provide for the protection of aquatic life and human health (USEPA 2013b). The following discussion of nutrients compares sampling results to these USEPA criteria. However, it is important to note that these criteria are established for freshwater systems, and as such, are only applicable to the freshwater portions of the Estuary. Currently, there are no numeric nutrient criteria established specifically for estuaries. However, Jenner will be included in the discussion for comparative purposes.

The USEPA desired goal for total nitrogen in Aggregate Ecoregion III is 0.38 mg/L for rivers and streams not discharging into lakes or reservoirs (USEPA 2000). Calculating total nitrogen values requires the summation of the different components of total nitrogen; organic and ammoniacal nitrogen (together referred to as Total Kjeldahl Nitrogen or TKN), and nitrate/nitrite nitrogen (Appendix 4.5).

Total nitrogen concentrations were only observed to exceed the recommended USEPA levels twice at the freshwater monitoring stations (Tables 4.1.2 through 4.1.6). Overall, total nitrogen exceedances constituted 1.9% of all freshwater samples collected (Figure 4.1.40). One exceedance occurred at Patterson Point on 9 June, and the other occurred at Vacation Beach on 16 June (Figure 4.1.40). The exceedance at Patterson Point occurred during closed conditions with a flow of approximately 124 cfs. The Vacation Beach exceedance occurred during open conditions with a flow of approximately 117 cfs. Patterson Point had a concentration of 0.40 mg/L (Table 4.1.4), while Vacation Beach had a concentration of 0.47 mg/L (Table 4.1.6). Whereas some of the lowest total nitrogen values observed at the freshwater stations occurred during closed conditions in September and October when flows were as low as 64 cfs (Figure 4.1.40). In contrast, the Jenner Station was observed to have several exceedances throughout the monitoring season during open and closed conditions with flows that ranged from 68 cfs to 183 cfs (Table 4.1.2).

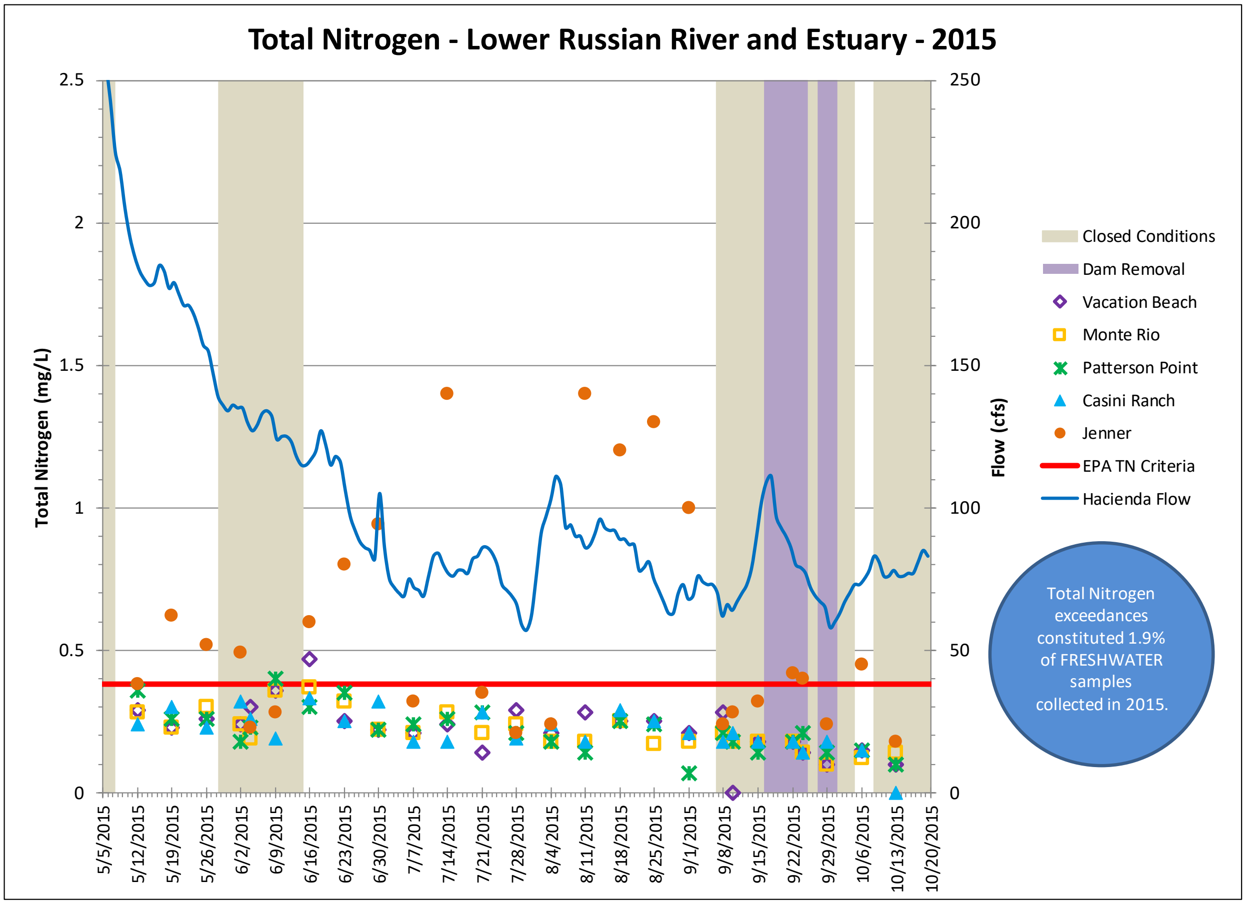


Figure 4.1.40. 2015 Russian River Grab Sampling Results for Total Nitrogen

The USEPA's desired goal for total phosphates as phosphorus in Aggregate Ecoregion III has been established as 21.88 micrograms per liter (µg/L), or approximately 0.022 mg/L, for rivers and streams not discharging into lakes or reservoirs (USEPA, 2000). Total phosphorus concentrations at the freshwater monitoring stations exceeded the U.S. EPA criteria approximately 86.5% of the time, continuing a trend of consistent exceedances observed in previous years. The maximum total phosphorus values recorded were 0.047 mg/L on June 16 at Casini Ranch, 0.064mg/L on 16 June at Patterson Point, 0.050 mg/L on June 16 at Monte Rio, and 0.042 mg/L on July 7 at Vacation Beach (Tables 4.1.3 through 4.1.6). The Jenner station was also observed to have several exceedances including a maximum recorded value of 0.065 mg/L on 12 May (Table 4.1.2). Interestingly, none of the stations exceeded the criteria for Total Phosphorus on September 8 when flows were only 62 cfs and the estuary had just closed the day before (Figure 4.1.41). Exceedances occurred in fresh and brackish water, during open and closed Estuary conditions, and in river flows ranging from 64 cfs to 183 cfs. Total phosphorus values were observed to generally be higher in the spring and early summer, trending downward through the rest of the season (Figure 4.1.41).

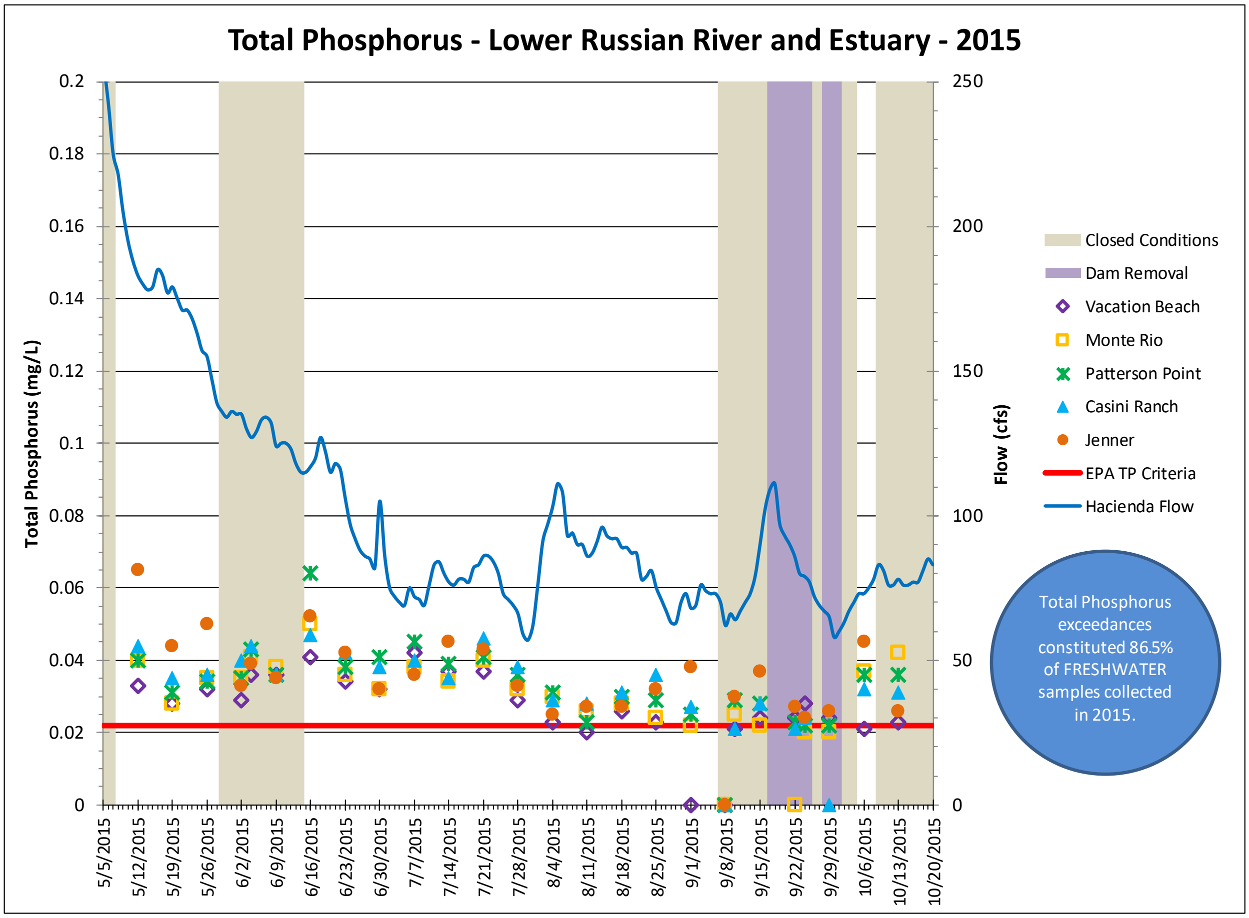


Figure 4.1.41. 2015 Russian River Grab Sampling Results for Total Phosphorus

Turbidity

There were no exceedances of the Turbidity EPA criteria at the Monte Rio and Casini Ranch stations and there were only two exceedances each at the Vacation Beach and Patterson Point stations (Figure 4.1.42). There were also several exceedances of the Turbidity criteria at Jenner under open and closed conditions in flows that ranged from 68 cfs to 183 cfs. Most exceedances were slightly higher than the EPA criteria of 2.34 NTU.

Chlorophyll a

In the process of photosynthesis, chlorophyll a (a green pigment in plants) absorbs sunlight and combines carbon dioxide and water to produce sugar and oxygen. Chlorophyll a can therefore serve as a measureable parameter of algal growth. Qualitative assessment of primary production on water quality can be based on chlorophyll a concentrations. A U.C. Davis report on the Klamath River (1999) assessing potential water quality and quantity regulations for restoration and protection of anadromous fish in the Klamath River includes a discussion of chlorophyll a and how it can affect water quality. The report characterizes the effects of chlorophyll a in terms of different levels of discoloration (e.g., no discoloration to some, deep, or very deep discoloration). The report indicated that less than 10 µg/L (or 0.01 mg/L) of chlorophyll a exhibits no discoloration (Deas and Orlob 1999). Additionally, the USEPA criterion for chlorophyll a in Aggregate Ecoregion III is 1.78 µg/L, or approximately 0.0018 mg/L for rivers

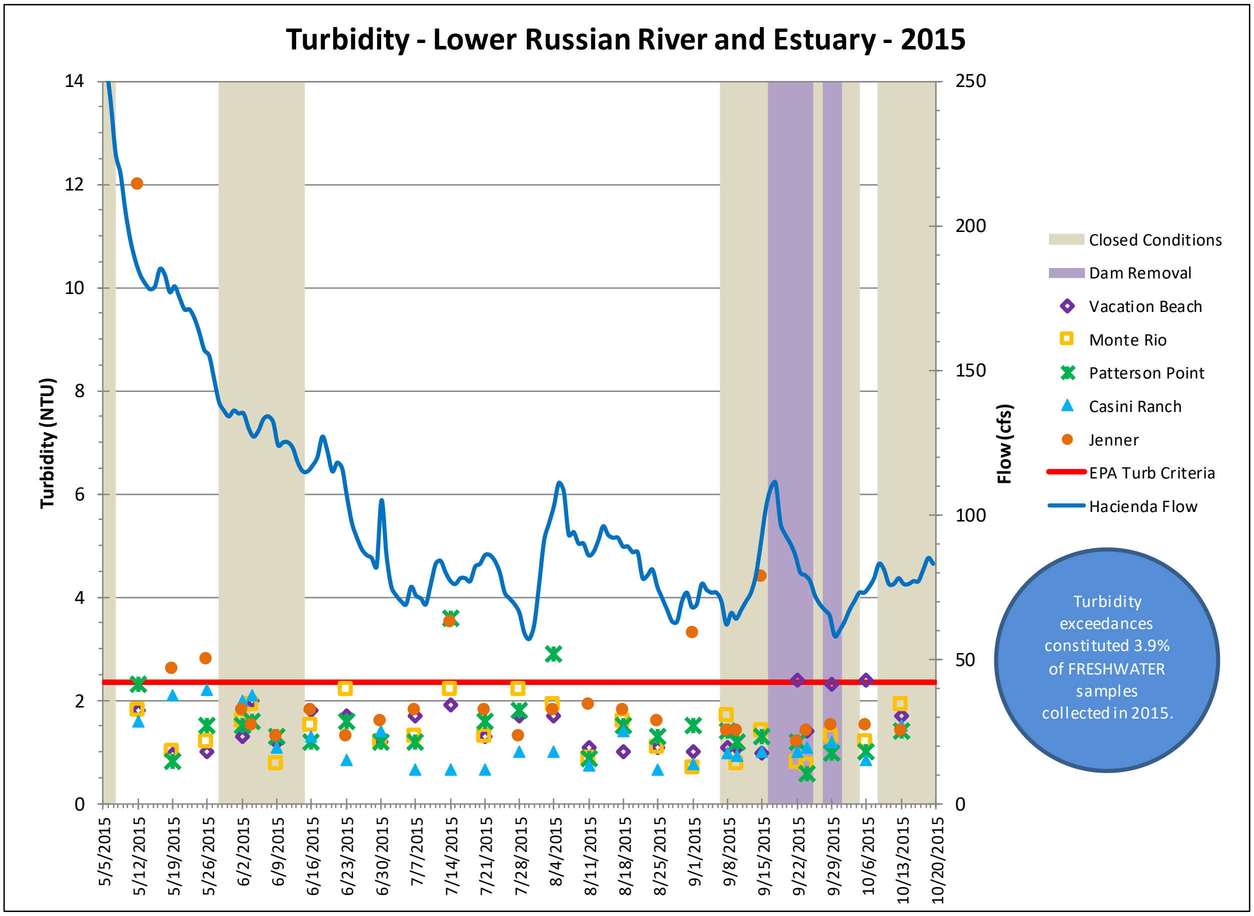


Figure 4.1.42. 2015 Russian River Grab Sampling Results for Turbidity

and streams not discharging into lakes or reservoirs (USEPA 2000). However, it is important to note that the USEPA criterion is established for freshwater systems, and as such, is only applicable to the freshwater portions of the Estuary. Currently, there are no numeric chlorophyll a criteria established specifically for estuaries.

Chlorophyll a concentrations were less than 0.01 mg/L at all stations during the monitoring period, the level recommended to prevent discoloration of surface waters, with the exception of one sampling event at the Jenner station (Tables 4.1.2 through 4.1.6). This sampling event occurred on 9 June with a chlorophyll a concentration of 0.011 mg/L (Table 4.1.2).

Chlorophyll a results exceeded the USEPA criteria approximately 26.0% of the time at the freshwater stations throughout the season in fresh and brackish water, under open and closed Estuary conditions, and during flows ranging from 62 cfs to 179 cfs (Figure 4.1.43). The maximum chlorophyll a concentrations were 0.0028 mg/L at the Casini Ranch station on 2 June, 0.0022 mg/L at the Patterson Point station on 7 July, 0.0025 mg/L at the Monte Rio station on 7 July, and 0.0034 mg/L at the Vacation Beach station on 7 July (Tables 4.1.2 through 4.1.6). Chlorophyll a concentrations were more pronounced at the Jenner station with a maximum chlorophyll a concentration of 0.011 mg/L recorded on 9 June, but again, this is an estuarine station and the USEPA criteria only apply to freshwater conditions (Figure 4.1.43).

Table 4.1.2. 2015 Jenner Station Grab Sample Results

Jenner Boat Ramp*	Temperature	Total Nitrogen	Phosphorus, Total	Turbidity	Chlorophyll-a	Total Coliforms (Collilert)	Total Coliforms Diluted 1:10 (Collilert)	E. coli (Collilert)	E. coli Diluted 1:10 (Collilert)	Enterococcus (Enterolert)	USGS 11467000 RR near Guerneville (Hacienda)***	
MDL**			0.020	0.020	0.000050	2	20	2	20	2	Flow Rate	Estuary
Date	°C	mg/L	mg/L	NTU	mg/L	MPN/100mL	MPN/100mL	MPN/100mL	MPN/100mL	MPN/100mL	(cfs)	Condition
5/12/2015	16.7	**0.38**	**0.065**	**12**	0.0015	>2419.6	2481.0	**1732.9**	**1956**	**435.2**	183	Open
5/19/2015	17.7	**0.62**	**0.044**	**2.6**	**0.0059**	>2419.6	583.0	12.1	31	6.3	179	Open
5/26/2015	17.1	**0.52**	**0.050**	**2.8**	**0.0074**	>2419.6	2142.0	9.7	10	3.0	155	Open
6/2/2015	18.0	**0.49**	**0.033**	1.8	**0.0027**	>2419.6	3876.0	24.3	50	58.3	135	Closed
6/4/2015	18.1	0.23	**0.039**	1.5	**0.0023**	>2419.6	1789	**290.9**	183	**98.5**	127	Closed
6/9/2015	20.0	0.28	**0.035**	1.3	**0.011**	1299.7	1539	93.3	121	24.3	124	Closed
6/16/2015	20.2	**0.60**	**0.052**	1.8	0.00047	>2419.6	**>24196**	2.0	10	**816.4**	117	Open
6/23/2015	17.7	**0.80**	**0.042**	1.3	0.0014	>2419.6	3076	3.0	<10	35.5	106	Open
6/30/2015	19.2	**0.94**	**0.032**	1.6	**0.0022**	>2419.6	**>24196**	45.9	122	**290.9**	105	Open
7/7/2015	19.4	0.32	**0.036**	1.8	**0.0044**	>2419.6	**>24196**	98.3	<10	31.3	72	Open
7/14/2015	20.0	**1.4**	**0.045**	**3.5**	**0.0031**	>2419.6	**12033**	31.8	<10	**261.3**	77	Open
7/21/2015	20.3	0.35	**0.043**	1.8	**0.0024**	>2419.6	**17329**	32.7	10	33.7	86	Open
7/28/2015	18.9	0.21	**0.033**	1.3	**0.0058**	>2419.6	**>24196**	**>2419.6**	20	**1046.2**	66	Open
8/4/2015	19.5	0.24	**0.025**	1.8	**0.0029**	>2419.6	**24196**	**1203.3**	109	**1299.7**	103	Open
8/11/2015	19.8	**1.4**	**0.027**	1.9	**0.0033**	>2419.6	**12033**	85.1	62	**1413.6**	86	Open
8/18/2015	18.8	**1.2**	**0.027**	1.8	**0.0021**	>2419.6	**19863**	**>2419.6**	86	**2419.6**	89	Open
8/25/2015	18.2	**1.3**	**0.032**	1.6	**0.0039**	>2419.6	**11199**	**>2419.6**	86	**920.8**	75	Open
9/1/2015	19.3	**1.0**	**0.038**	**3.3**	**0.0024**	>2419.6	6488.0	**866.4**	86	**410.6**	68	Open
9/8/2015	17.4	0.24	ND	1.4	**0.0060**	>2419.6	2723.0	**387.3**	121	**1725.0**	62	Closed
9/10/2015	17.8	0.28	**0.030**	1.4	**0.0082**	1732.9	402.0	**290.9**	10	**88.6**	64	Closed
9/15/2015	16.6	0.32	**0.037**	**4.4**	**0.0049**	>2419.6	**12033.0**	**281.2**	20	**178.5**	90	Closed
9/22/2015	19.1	**0.42**	**0.027**	1.2	**0.0042**	>2419.6	583.0	26.6	41	28.8	86	Closed
9/24/2015	18.0	**0.40**	**0.024**	1.4	**0.0031**	>2419.6	1597.0	65.7	63	**150.0**	79	Closed
9/29/2015	18.5	0.24	**0.026**	1.5	**0.0051**	648.8	285.0	6.3	<10	8.5	65	Closed
10/6/2015	19.4	**0.45**	**0.045**	1.5	0.0015	>2419.6	**19863.0**	11.0	<10	48.5	73	Open
10/13/2015	17.6	0.18	**0.026**	1.4	**0.0023**	>2419.6	**>24196**	**325.5**	**256**	**>2419.6**	78	Closed

* All results are preliminary and subject to final revision

** Method Detection Limit - limits can vary for individual samples depending on matrix interference and dilution factors.

*** United States Geological Survey (USGS) Continuous-Record Gaging Station (Flow rates are preliminary and subject to final revision by USGS).

Recommended EPA Criteria based on Aggregate Ecoregion III
Total Phosporus: 0.02188 mg/L (21.88 ug/L) ≈ 0.022 mg/L
Total Nitrogen: 0.38 mg/L
Chlorophyll *a* : 0.00178 mg/L (1.78 ug/L) ≈ 0.0018 mg/L
Turbidity: 2.34 FTU/NTU

CDPH Draft Guidance for Fresh Water Beaches - Single Sample Values:
Beach posting is recommended when indicator organisms exceed any of the following levels:
Total coliforms: 10,000 per 100 ml
E. coli: 235 per 100 ml
Enterococcus: 61 per 100 ml

Table 4.1.3. 2015 Casini Ranch Station Grab Sample Results

Casini Ranch*	Temperature	Total Nitrogen	Phosphorus, Total	Turbidity	Chlorophyll-a	Total Coliforms (Collilert)	Total Coliforms Diluted 1:10 (Collilert)	E. coli (Collilert)	E. coli Diluted 1:10 (Collilert)	Enterococcus (Enterolert)	USGS 11467000 RR near Guerneville (Hacienda)***	
MDL**			0.020	0.020	0.000050	2	20	2	20	2	Flow Rate	Estuary
Date	°C	mg/L	mg/L	NTU	mg/L	MPN/100mL	MPN/100mL	MPN/100mL	MPN/100mL	MPN/100mL	(cfs)	Condition
5/12/2015	20.1	0.24	**0.044**	1.6	0.0015	547.5	677	5.2	<10	2.0	183	Open
5/19/2015	20.4	0.30	**0.035**	2.1	0.0013	816.4	749	22.8	10	5.2	179	Open
5/26/2015	20.6	0.23	**0.036**	2.2	**0.0027**	686.7	932	6.3	<10	8.5	155	Open
6/2/2015	21.5	0.32	**0.040**	2.0	**0.0028**	1299.7	1607	27.9	75	47.4	135	Closed
6/4/2015	21.2	0.26	**0.044**	2.1	**0.0024**	1553.1	1720	47.1	98	35.5	127	Closed
6/9/2015	22.8	0.19	**0.036**	1.1	0.0016	1732.9	1354	43.5	31	25.6	124	Closed
6/16/2015	22.3	0.33	**0.047**	1.3	0.00082	>2419.6	2489	8.4	<10	2.0	117	Open
6/23/2015	22.2	0.25	**0.042**	0.85	**0.0021**	2419.6	2014	6.3	10	7.3	106	Open
6/30/2015	23.6	0.32	**0.038**	1.4	0.0012	>2419.6	7270	15.8	31	7.4	105	Open
7/7/2015	23.1	0.18	**0.040**	0.66	0.0014	>2419.6	**11199**	7.4	10	2.0	72	Open
7/14/2015	24.0	0.18	**0.035**	0.65	0.0013	2419.6	1860	8.4	<10	16.0	77	Open
7/21/2015	24.8	0.28	**0.046**	0.66	0.0012	2419.6	1421	4.1	20	3.1	86	Open
7/28/2015	23.4	0.19	**0.038**	1.0	0.0009	1119.9	960	5.1	20	9.6	66	Open
8/4/2015	22.7	0.24	**0.029**	1.0	0.0014	770.1	809	4.1	10	1.0	103	Open
8/11/2015	23.1	0.18	**0.028**	0.75	0.00064	1299.7	1100	6.2	<10	4.1	86	Open
8/18/2015	22.3	0.29	**0.031**	1.4	0.00074	1119.9	767	5.2	<10	2.0	89	Open
8/25/2015	21.3	0.25	**0.036**	0.67	0.00094	816.4	851	14.6	10	3.1	75	Open
9/1/2015	23.5	0.21	**0.027**	0.78	0.0012	816.4	689	8.6	<10	2.0	68	Open
9/8/2015	21.5	0.18	ND	0.98	0.00096	920.8	884	7.4	10	41.0	62	Closed
9/10/2015	21.7	0.21	0.021	0.92	0.0011	980.4	620	13.4	20	3.1	64	Closed
9/15/2015	21.2	0.18	**0.028**	1.0	**0.0019**	1413.6	1664	38.4	75	60.2	90	Closed
9/22/2015	21.7	0.18	0.021	1.0	**0.0019**	1413.6	1354	42.2	63	45.0	86	Closed
9/24/2015	20.0	0.14	**0.024**	1.1	0.0015	1986.3	1956	60.2	63	**79.4**	79	Closed
9/29/2015	20.1	0.18	ND	1.2	**0.0021**	1119.9	1314	42.0	75	**82.0**	65	Closed
10/6/2015	19.4	0.15	**0.032**	0.84	0.0013	547.5	512	14.5	20	6.3	73	Open
10/13/2015	20.0	ND	**0.031**	1.5	0.00071	1986.3	2143	28.1	74	58.1	78	Closed

* All results are preliminary and subject to final revision

** Method Detection Limit - limits can vary for individual samples depending on matrix interference and dilution factors.

*** United States Geological Survey (USGS) Continuous-Record Gaging Station (Flow rates are preliminary and subject to final revision by USGS).

Recommended EPA Criteria based on Aggregate Ecoregion III

Total Phosporus: 0.02188 mg/L (21.88 ug/L) ≈ 0.022 mg/L

Total Nitrogen: 0.38 mg/L

Chlorophyll *a* : 0.00178 mg/L (1.78 ug/L) ≈ 0.0018 mg/L

Turbidity: 2.34 FTU/NTU

CDPH Draft Guidance for Fresh Water Beaches - Single Sample Values:

Beach posting is recommended when indicator organisms exceed any of the following levels:

Total coliforms: 10,000 per 100 ml

E. coli: 235 per 100 ml

Enterococcus: 61 per 100 ml

Table 4.1.4. 2015 Patterson Point Station Grab Sample Results

Patterson Point*	Temperature	Total Nitrogen	Phosphorus, Total	Turbidity	Chlorophyll-a	Total Coliforms (Collilert)	Total Coliforms Diluted 1:10 (Collilert)	E. coli (Collilert)	E. coli Diluted 1:10 (Collilert)	Enterococcus (Enterolert)	USGS 11467000 RR near Guerneville (Hacienda)***	
MDL**			0.020	0.020	0.000050	2	20	2	20	2	Flow Rate	Estuary
Date	°C	mg/L	mg/L	NTU	mg/L	MPN/100mL	MPN/100mL	MPN/100mL	MPN/100mL	MPN/100mL	(cfs)	Condition
5/12/2015	19.5	0.36	**0.040**	2.3	0.0011	770.1	521	4.1	10	3.1	183	Open
5/19/2015	20.0	0.26	**0.031**	0.82	0.00083	547.5	512	14.8	20	6.3	179	Open
5/26/2015	20.6	0.26	**0.034**	1.5	**0.0019**	770.1	1050	14.6	10	7.3	155	Open
6/2/2015	20.3	0.18	**0.035**	1.5	0.0016	1046.2	906	26.2	10	32.7	135	Closed
6/4/2015	21.0	0.23	**0.043**	1.6	0.0010	1299.7	1674	32.7	10	49.6	127	Closed
6/9/2015	23.6	**0.40**	**0.036**	1.3	0.00082	1732.9	2481	36.9	41	22.8	124	Closed
6/16/2015	22.5	0.30	**0.064**	1.2	0.00082	>2419.6	4352	20.1	30	20.0	117	Open
6/23/2015	22.7	0.35	**0.038**	1.6	**0.0021**	2419.6	1722	5.2	<10	18.7	106	Open
6/30/2015	23.5	0.22	**0.041**	1.2	**0.0018**	1553.1	2603	39.9	20	16.9	105	Open
7/7/2015	23.7	0.24	**0.045**	1.2	**0.0022**	>2419.6	2909	12.2	41	14.1	72	Open
7/14/2015	23.8	0.26	**0.039**	**3.6**	0.0014	1986.3	1904	37.3	31	42.5	77	Open
7/21/2015	24.8	0.28	**0.041**	1.6	0.00094	1986.3	2143	6.3	10	4.1	86	Open
7/28/2015	24.1	0.21	**0.036**	1.8	0.0016	1046.2	1872	52.0	52	6.3	66	Open
8/4/2015	23.5	0.18	**0.031**	**2.9**	0.00091	1553.1	2187	5.2	10	12.8	103	Open
8/11/2015	23.2	0.14	**0.023**	0.88	0.0013	1553.1	2143	6.3	<10	3.1	86	Open
8/18/2015	23.2	0.25	**0.030**	1.5	0.00050	1553.1	2046	4.1	10	7.4	89	Open
8/25/2015	22.1	0.24	**0.029**	1.3	0.00094	920.8	1145	17.5	<10	19.9	75	Open
9/1/2015	23.5	0.070	**0.025**	1.5	0.0011	472.1	1081	8.6	20		68	Open
9/8/2015	21.9	0.21	ND	1.4	0.00068	770.1	749	5.2	31	10.0	62	Closed
9/10/2015	22.1	0.18	**0.029**	1.2	0.0016	866.4	1198	9.0	<10	8.4	64	Closed
9/15/2015	20.8	0.14	**0.028**	1.3	**0.0019**	2419.6	2046	69.1	74	26.5	90	Closed
9/22/2015	21.0	0.18	**0.023**	1.2	0.0013	1299.7	1333	96.0	98	**95.9**	86	Closed
9/24/2015	20.4	0.21	**0.022**	0.58	0.00093	1553.1	1860	63.7	85	**93.3**	79	Closed
9/29/2015	19.8	0.14	**0.022**	0.99	0.0015	613.1	1236	42.0	20	**62.0**	65	Closed
10/6/2015	20.0	0.15	**0.036**	1.0	0.00087	816.4	813	14.5	20	27.5	73	Open
10/13/2015	19.3	0.10	**0.036**	1.4	0.0011	1203.3	1291	68.3	**331**	59.4	78	Closed

* All results are preliminary and subject to final revision

** Method Detection Limit - limits can vary for individual samples depending on matrix interference and dilution factors.

*** United States Geological Survey (USGS) Continuous-Record Gaging Station (Flow rates are preliminary and subject to final revision by USGS).

Recommended EPA Criteria based on Aggregate Ecoregion III

Total Phosporus: 0.02188 mg/L (21.88 ug/L) ≈ 0.022 mg/L

Total Nitrogen: 0.38 mg/L

Chlorophyll *a* : 0.00178 mg/L (1.78 ug/L) ≈ 0.0018 mg/L

Turbidity: 2.34 FTU/NTU

CDPH Draft Guidance for Fresh Water Beaches - Single Sample Values:

Beach posting is recommended when indicator organisms exceed any of the following levels:

Total coliforms: 10,000 per 100 ml

E. coli: 235 per 100 ml

Enterococcus: 61 per 100 ml

Table 4.1.5. 2015 Monte Rio Station Grab Sample Results

Monte Rio*	Temperature	Total Nitrogen	Phosphorus, Total	Turbidity	Chlorophyll-a	Total Coliforms (Colilert)	Total Coliforms Diluted 1:10 (Colilert)	E. coli (Colilert)	E. coli Diluted 1:10 (Colilert)	Enterococcus (Enterolert)	USGS 11467000 RR near Guerneville (Hacienda)***	
MDL**			0.020	0.020	0.000050	2	20	2	20	2	Flow Rate	Estuary
Date	°C	mg/L	mg/L	NTU	mg/L	MPN/100mL	MPN/100mL	MPN/100mL	MPN/100mL	MPN/100mL	(cfs)	Condition
5/12/2015	19.5	0.28	**0.040**	1.8	0.0014	727	880	8.5	20	5.2	183	Open
5/19/2015	20.1	0.23	**0.028**	1.0	0.0012	920.8	697	14.6	<10	1.0	179	Open
5/26/2015	20.8	0.30	**0.035**	1.2	**0.0019**	686.7	1145	13.4	10	3.0	155	Open
6/2/2015	20.4	0.24	**0.035**	1.6	0.0010	866.4	1274	22.8	10	6.3	135	Closed
6/4/2015	21.3	0.19	**0.041**	1.9	0.00028	913.9	2181	67.6	110	45.7	127	Closed
6/9/2015	23.7	0.36	**0.038**	0.77	0.0011	>2419.6	2613	76.7	121	48.7	124	Closed
6/16/2015	22.4	0.37	**0.050**	1.5	0.00070	>2419.6	5172	43.5	20	37.3	117	Open
6/23/2015	23.2	0.32	**0.036**	2.2	**0.0023**	1732.9	3448	31.3	20	13.1	106	Open
6/30/2015	24.5	0.22	**0.032**	1.2	0.0012	1046.2	1607	20.1	10	4.1	105	Open
7/7/2015	23.6	0.21	**0.038**	1.3	**0.0025**	1553.1	2909	18.1	98	17.4	72	Open
7/14/2015	23.6	0.28	**0.034**	2.2	0.0015	1732.9	2909	13.1	<10	36.8	77	Open
7/21/2015	25.0	0.21	**0.040**	1.3	**0.0019**	1413.6	2187	6.3	41	3.0	86	Open
7/28/2015	23.7	0.24	**0.032**	2.2	0.0014	1553.1	1597	12.0	20	22.8	66	Open
8/4/2015	23.9	0.18	**0.030**	1.9	0.0011	1986.3	1670	9.8	10	20.6	103	Open
8/11/2015	23.5	0.18	**0.026**	0.88	0.0010	1299.7	1223	2.1	<10	6.2	86	Open
8/18/2015	23.8	0.25	**0.028**	1.6	0.00074	1986.3	1421	14.6	20	5.2	89	Open
8/25/2015	22.0	0.17	**0.024**	1.1	**0.0020**	1119.9	1119	5.2	<10	5.2	75	Open
9/1/2015	23.5	0.18	**0.022**	0.70	0.0011	980.4	882	3.1	<10	2.0	68	Open
9/8/2015	21.8	0.21	ND	1.7	0.0014	920.8	959	7.3	20	41.0	62	Closed
9/10/2015	21.6	0.18	**0.025**	0.77	0.0011	727.0	1198	7.5	<10	3.0	64	Closed
9/15/2015	20.2	0.18	**0.022**	1.4	0.0014	1046.2	1450	6.2	<10	7.4	90	Closed
9/22/2015	21.4	0.18	ND	0.79	0.00080	1986.3	1374	58.3	62	**98.7**	86	Closed
9/24/2015	20.3	0.14	0.020	0.73	0.00053	1986.3	1515	70.6	63	**93.3**	79	Closed
9/29/2015	20.4	0.10	0.020	1.3	0.0011	2419.6	1439	**307.6**	110	**98.8**	65	Closed
10/1/2015	----	----	----	----	----	913.9	1932	97.7	41	**80.5**	59	Closed
10/6/2015	19.6	0.12	**0.037**	1.2	0.00087	1203.3	1376	15.8	<10	27.5	73	Open
10/13/2015	19.4	0.14	**0.042**	1.9	0.0014	980.4	624	12.1	<10	11.0	78	Closed

* All results are preliminary and subject to final revision

** Method Detection Limit - limits can vary for individual samples depending on matrix interference and dilution factors.

*** United States Geological Survey (USGS) Continuous-Record Gaging Station (Flow rates are preliminary and subject to final revision by USGS).

Recommended EPA Criteria based on Aggregate Ecoregion III

Total Phosporus: 0.02188 mg/L (21.88 ug/L) ≈ 0.022 mg/L

Total Nitrogen: 0.38 mg/L

Chlorophyll *a* : 0.00178 mg/L (1.78 ug/L) ≈ 0.0018 mg/L

Turbidity: 2.34 FTU/NTU

CDPH Draft Guidance for Fresh Water Beaches - Single Sample Values:

Beach posting is recommended when indicator organisms exceed any of the following levels:

Total coliforms: 10,000 per 100 ml

E. coli: 235 per 100 ml

Enterococcus: 61 per 100 ml

Table 4.1.6. 2015 Vacation Beach Station Grab Sample Results

Vacation Beach*	Temperature	Total Nitrogen**	Phosphorus, Total	Turbidity	Chlorophyll-a	Total Coliforms (Colilert)	Total Coliforms Diluted 1:10 (Colilert)	E. coli (Colilert)	E. coli Diluted 1:10 (Colilert)	Enterococcus (Enterolert)	USGS 11467000 RR near Guerneville (Hacienda)***	
MDL**			0.020	0.020	0.000050	2	20	2	20	2	Flow Rate	Estuary
Date	°C	mg/L	mg/L	NTU	mg/L	MPN/100mL	MPN/100mL	MPN/100mL	MPN/100mL	MPN/100mL	(cfs)	Condition
5/12/2015	19.5	0.29	**0.033**	1.8	0.0015	722	789	12.1	10	<1.0	183	Open
5/19/2015	20.2	0.23	**0.028**	0.96	**0.0018**	727.0	697	7.5	10	13.0	179	Open
5/26/2015	21.1	0.26	**0.032**	1.0	0.0017	613.1	1019	10.9	10	8.6	155	Open
6/2/2015	20.8	0.24	**0.029**	1.3	0.0010	920.8	1314	21.8	10	16.1	135	Closed
6/4/2015	21.2	0.30	**0.036**	2.0	0.0013	866.4	1935	27.2	10	21.3	127	Closed
6/9/2015	23.7	0.36	**0.036**	1.2	0.00082	1208.3	1565	10.9	10	30.8	124	Closed
6/16/2015	22.9	**0.47**	**0.041**	1.8	0.0015	2419.6	5475	45.0	41	**73.3**	117	Open
6/23/2015	23.1	0.25	**0.034**	1.7	**0.0031**	>2419.6	**19863**	41.4	<10	54.6	106	Open
6/30/2015	24.6	0.22	**0.032**	1.2	**0.0019**	>2419.6	**11199**	21.8	41	22.6	105	Open
7/7/2015	24.0	0.21	**0.042**	1.7	**0.0034**	>2419.6	5475	14.6	30	52.1	72	Open
7/14/2015	23.7	0.24	**0.037**	1.9	**0.0024**	2419.6	2481	24.6	10	14.6	77	Open
7/21/2015	25.2	0.14	**0.037**	1.3	**0.0028**	>2419.6	3448	63.7	98	47.1	86	Open
7/28/2015	24.5	0.29	**0.029**	1.7	0.0016	>2419.6	2481	17.3	20	**204.6**	66	Open
8/4/2015	24.1	0.21	**0.023**	1.7	0.0016	>2419.6	4106	9.6	10	38.9	103	Open
8/11/2015	23.7	0.28	0.020	1.1	0.0010	2419.6	1860	2.0	<10	16.0	86	Open
8/18/2015	23.9	0.25	**0.026**	1.0	**0.0020**	1732.9	2755	23.1	<10	45	89	Open
8/25/2015	22.3	0.25	**0.023**	1.1	**0.0023**	1413.6	1624	8.3	<10	9.5	75	Open
9/1/2015	23.9	0.21	ND	1.0	**0.0020**	1986.3	1872	4.1	10	6.3	68	Open
9/8/2015	21.9	0.28	ND	1.1	0.0015	1986.3	1723	1.0	10	**63.0**	62	Closed
9/10/2015	22.0	ND	0.021	1.1	**0.0019**	1732.9	2755	10.9	10	8.6	64	Closed
9/15/2015	20.8	0.18	**0.024**	0.99	0.0015	2419.6	1785	48.7	41	20.1	90	Closed
9/22/2015	21.0	0.18	**0.024**	**2.4**	0.00080	1203.3	1081	30.5	52	16.0	86	Closed
9/24/2015	20.1	0.14	**0.028**	1.4	0.00080	960.6	1187	51.2	73	**76.7**	79	Closed
9/29/2015	19.9	0.10	**0.024**	2.3	0.0016	1299.7	1670	114.5	146	**228.2**	65	Closed
10/1/2015	----	----	----	----	----	>2419.6	**>24196**	**>2419.6**	**7270**	**>2419.6**	59	Closed
10/6/2015	19.5	0.15	0.021	**2.4**	0.0016	980.4	1198	44.1	108	42.2	73	Open
10/13/2015	19.6	0.10	**0.023**	1.7	0.0013	980.4	1211	45.9	109	**85.5**	78	Closed

* All results are preliminary and subject to final revision

** Method Detection Limit - limits can vary for individual samples depending on matrix interference and dilution factors.

*** United States Geological Survey (USGS) Continuous-Record Gaging Station (Flow rates are preliminary and subject to final revision by USGS).

Recommended EPA Criteria based on Aggregate Ecoregion III

Total Phosporus: 0.02188 mg/L (21.88 ug/L) ≈ 0.022 mg/L

Total Nitrogen: 0.38 mg/L

Chlorophyll *a* : 0.00178 mg/L (1.78 ug/L) ≈ 0.0018 mg/L

Turbidity: 2.34 FTU/NTU

CDPH Draft Guidance for Fresh Water Beaches - Single Sample Values:

Beach posting is recommended when indicator organisms exceed any of the following levels:

Total coliforms: 10,000 per 100 ml

E. coli: 235 per 100 ml

Enterococcus: 61 per 100 ml

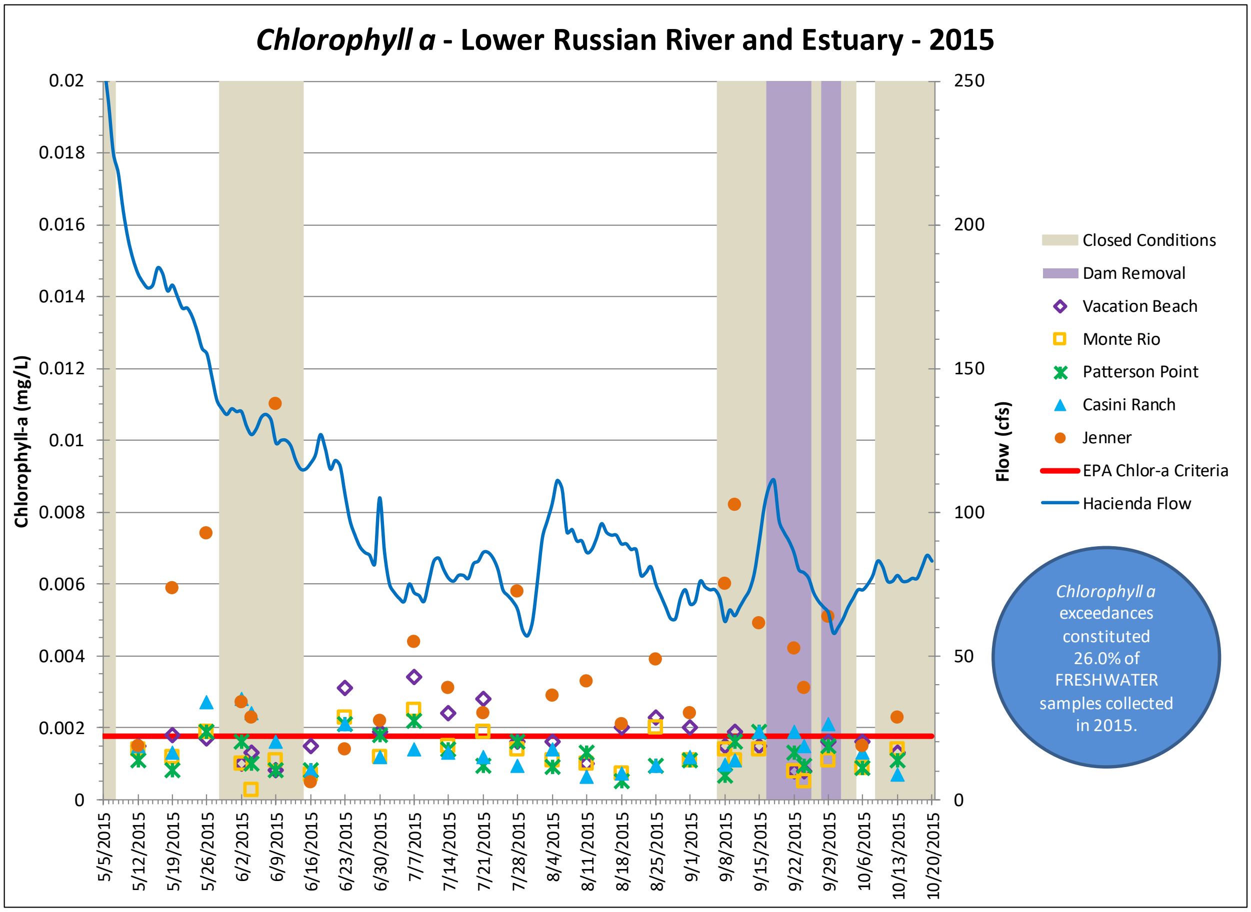


Figure 4.1.43. 2015 Russian River Grab Sampling Results for Chlorophyll a

Indicator Bacteria

The California Department of Public Health (CDPH) developed the "Draft Guidance for Fresh Water Beaches," which describes bacteria levels that, if exceeded, may require posted warning signs in order to protect public health (CDPH 2011). The CDPH draft guideline for single sample maximum concentrations is: 10,000 most probable numbers (MPN) per 100 milliliters (ml) for total coliform, 235 MPN per 100 ml for E. coli, and 61 MPN per 100 ml for Enterococcus. In 2012, the USEPA issued Clean Water Act (CWA) §304(a) Recreational Water Quality Criteria (RWQC) for States (USEPA 2012). The RWQC recommends using two criteria for assessing water quality relating to fecal indicator bacteria: the geometric mean (GM) of the dataset, and changing the single sample maximum (SSM) to a Statistical Threshold Value (STV) representing the 75th percentile of an acceptable water-quality distribution. However, the USEPA recommends using STV values as SSM values for potential recreational beach posting and those values are provided in this report for comparative purposes. It must be emphasized that these are draft guidelines and criteria, not adopted standards, and are therefore both subject to change (if it is determined that the guidelines and/or criteria are not accurate indicators) and are not currently enforceable. In addition, these draft guidelines and criteria were established for and are only applicable to fresh water beaches. Currently, there are no numeric guidelines or criteria that have been developed for estuarine areas. The Jenner Boat Ramp grab sample station is located in an area that is predominantly brackish water, whereas the four upstream grab sample stations are located in predominantly freshwater habitat (Casini Ranch, Patterson Point, Monte Rio, and Vacation Beach).

Samples were collected during the monitoring season for diluted and undiluted analysis of E. coli and total coliform for comparative purposes and the results are included in Tables 4.1.2 through 4.1.6 and Figures 4.1.44 and 4.1.45. Samples collected for Enterococcus were undiluted only and results are included in Tables 4.1.2 through 4.1.6 and Figure 4.1.46. The Water Agency submitted samples to the Sonoma County DHS Public Health Division Lab in Santa Rosa for bacteria analysis. E. coli and total coliform were analyzed using the Colilert method and Enterococcus was analyzed using the Enterolert method. Samples for all other constituents were submitted to Alpha Labs in Ukiah for analysis.

Following the 2015 monitoring season, Water Agency staff discovered issues with the reliability of bacteria data that has been collected in the presence of brackish water in the Estuary. In 2014, the Jenner station had a couple of anomalous results for undiluted samples of E. coli compared to diluted samples collected at the same time. In 2015 it was more significant and frequent, with undiluted E. coli results often being >2419.6 MPN, compared to a value of less than 100 MPN in the diluted sample.

Water Agency staff contacted Sonoma County Department of Health Services (DHS) to see if the high E. coli results for the undiluted samples at Jenner were errors. DHS staff responded and explained that marine waters can create false positives when relying on the Colilert analysis if the samples are not diluted (Ferris pers. comm.). DHS staff also stated that any samples collected in marine waters should be diluted at a one to ten ratio (1:10). Water Agency staff conducted additional literature research and discovered that other non-coliform bacteria commonly found in marine waters (as well as plant and algal material) can produce false positives for total coliforms and E. coli if not diluted when using the IDEXX Colilert analytical methodology (Pisciotta 2002). In addition, the IDEXX Colilert SOP states to dilute samples 1:10 if specific conductance is between 3,000 and 10,000 microsiemens (µs) and to not use the IDEXX Colilert at all if the samples are greater than 10,000 microsiemens (IDEXX 2015).

In the last three years, Water Agency staff have collected two (2) samples at Jenner when the water was less than 3,000 µs, out of 81 samples. The majority were over 10,000 µs. In 2015, 15 of 26 sample events at Jenner were in water with specific conductance values over 10,000 µs. In 2013 it was 15 of 29, and 2014 was 19 of 26.

DHS staff also stated that the Enterolert analysis could produce false positives in marine waters and a study conducted in Georgia observed saltwater interference with the Enterolert system and recommended that samples collected in marine waters should be diluted 1:10 to reduce the number of false positive results (McDonald 2003). Water Agency staff have been relying on Colilert and Enterolert since 2012, but only started having samples diluted for E. coli and total coliform in 2014 for part of the season, and in 2015 for all of the season. Enterococcus samples have not been diluted.

Essentially, the bacteria data collected at the Jenner station is predominantly unreliable due to the saline conditions at the site, although the diluted results for E. coli and total coliform did

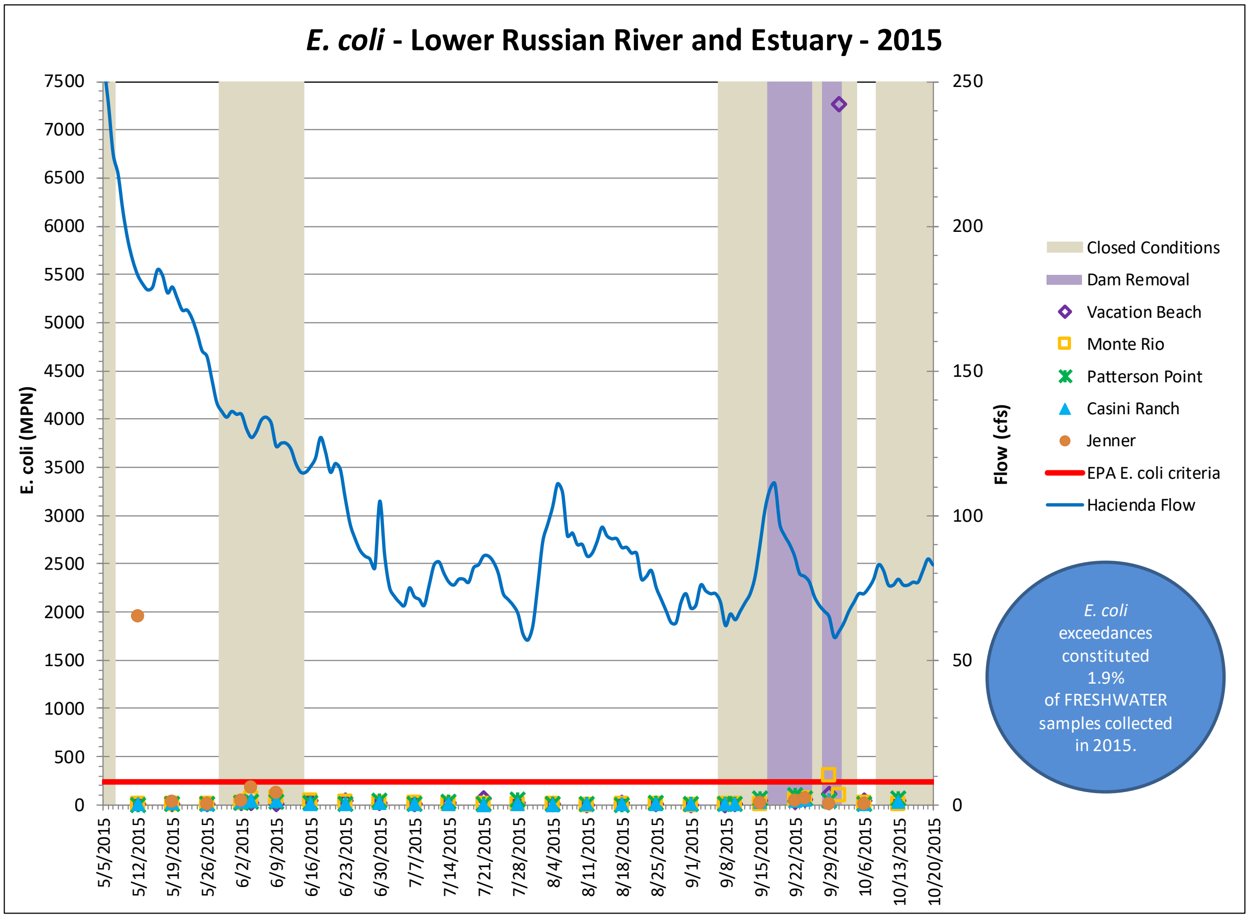


Figure 4.1.44. 2015 Russian River Grab Sampling Results for E. coli

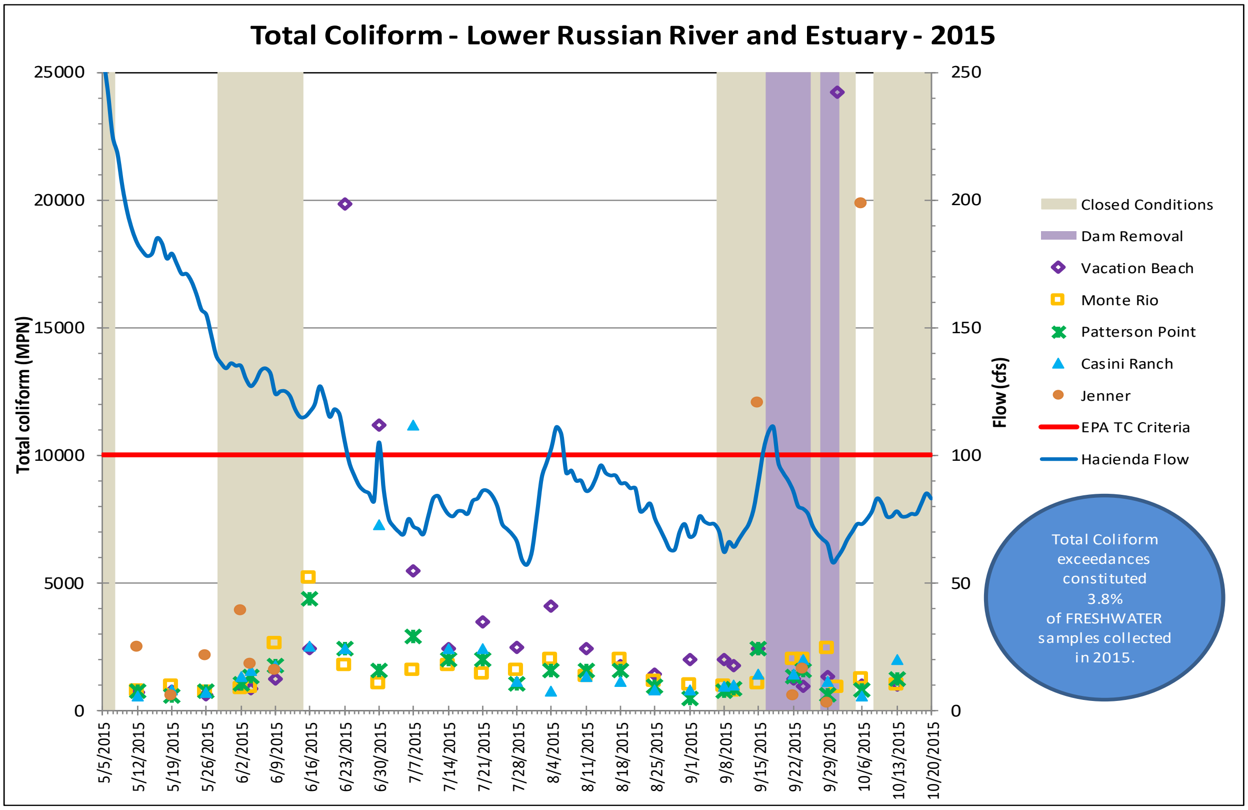


Figure 4.1.45. 2015 Russian River Grab Sampling Results for Total Coliform

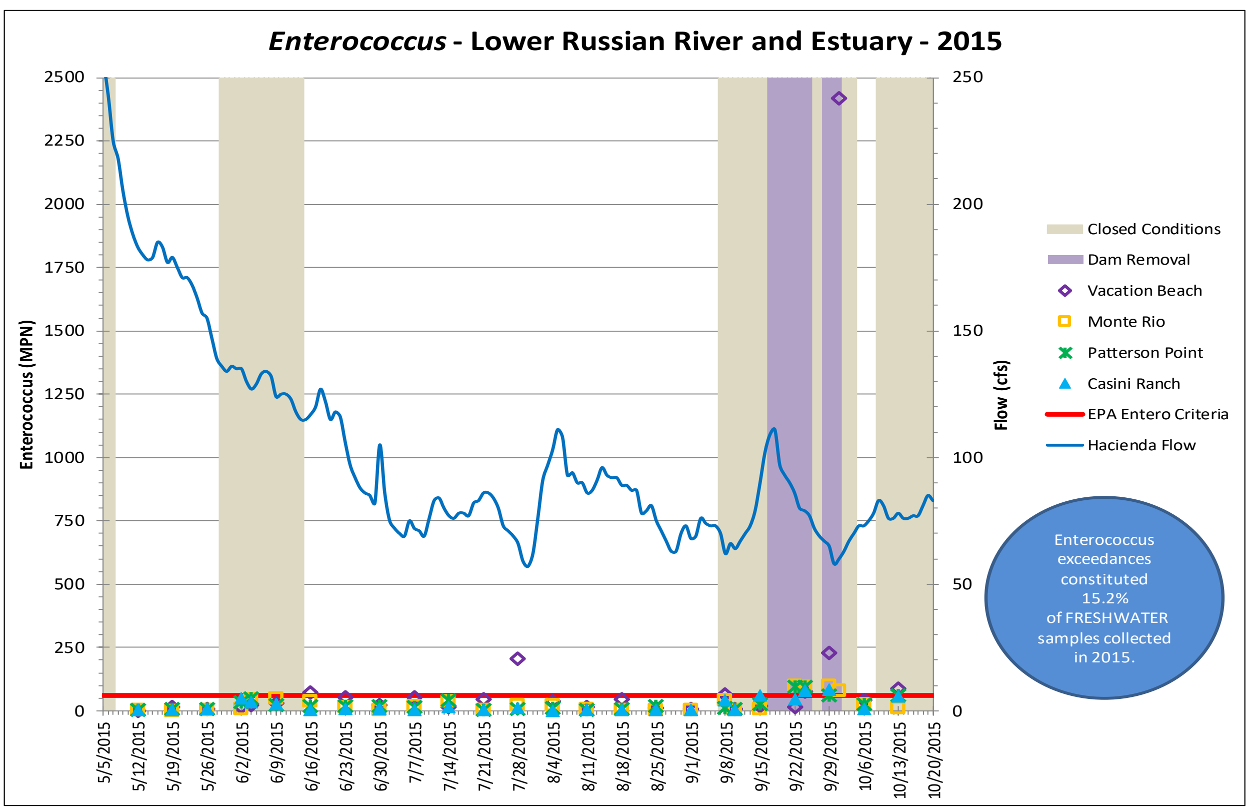


Figure 4.1.46. 2015 Russian River Grab Sampling Results for Enterococcus

include some results that were collected in water with specific conductance values below 10,000 μs and should be considered reliable and are included in Figures 4.1.44 and 4.1.45. Because the Enterococcus samples at Jenner were undiluted, results will not be included in Figure 4.1.46, but are included in Table 4.1.2. Finally, E. coli and total coliform data presented in Figures 4.1.44 and 4.1.45 utilize undiluted sample results unless the reporting limit has been exceeded, at which point the diluted results are utilized.

In 2014, staff at the NCRWQCB indicated that Enterococcus was not being utilized as a fecal indicator bacteria due to uncertainty in the validity of the lab analysis to produce accurate results, as well as evidence that Enterococcus colonies can be persistent in the water column and therefore its presence at a given site may not always be associated with a fecal source. Water Agency staff will continue to collect Enterococcus samples and record and report the data however, Enterococcus results will not be relied upon when coordinating with the NCRWQCB and Sonoma County DHS about potentially posting warning signs at freshwater beach sites or to discuss potential adaptive management actions including mechanical breaching of the sandbar to address potential threats to public health.

NCRWQCB staff also indicated during the 2014 monitoring season that they were uncertain of the validity of the laboratory analysis for *Bacteroides* and would not be conducting lab analysis of the samples until the question of validity had been resolved. As a result, Water Agency staff did not collect surface-water samples to test for *Bacteroides* during the 2015 monitoring season.

The Monte Rio and Vacation Beach stations were observed to have one exceedance each of the RWQC for E. coli following summer dam removal, representing 1.9% of the total freshwater samples collected (Figure 4.1.44). Whereas the exceedance at Monte Rio was slightly above the recommended criteria, the exceedance at Vacation Beach had a concentration of 7,270 MPN that was observed to occur on 1 October during the removal of the Johnson's Beach summer dam (Table 4.1.6). Jenner had one exceedance of the RWQC for E. coli during the term of the Order on May 12 during open conditions with a flow of 183 cfs (Table 4.1.2).

There were several exceedances of the RWQC for total coliform including three exceedances at Vacation Beach, two exceedances at Jenner and one exceedance at Casini Ranch (Figure 4.1.45). Total coliform exceedances, representing approximately 3.8% of freshwater samples collected, occurred during open and closed estuary conditions with flows that ranged from 59 cfs to 106 cfs (Tables 4.1.2 through 4.1.6). Summer dam removal may have had an effect on total coliform with a concentration of over 24,196 MPN observed to occur on October 1 during the removal of the Johnson's Beach summer dam (Figure 4.1.45).

Based upon the recommended Enterococcus RWQC for freshwater beaches, several exceedances were observed in the latter half of the season at the freshwater stations, with flows varying from 62 cfs to 86 cfs. External factors likely had an effect on increasing Enterococcus concentrations including the removal of the summer dams in Guerneville during an extended period of estuary closures. Similar to the E.coli and total coliform results, the Vacation Beach station was observed to have a concentration over 2419.6 (>2419.6) MPN that occurred during the removal of the Johnson's Beach summer dam on October 1 (Figure 4.1.46).

Conclusions and Recommendations

Continuous Water Quality Monitoring Conclusions

Water quality conditions observed during the 2015 monitoring season were similar to conditions observed during previous monitoring seasons, and similar to the dynamic conditions associated with an estuarine river system. The differing physical properties associated with freshwater versus those of saltwater play a pivotal role in the stratification that is common in the Russian River Estuary. Since the saltwater is denser than the freshwater inflow, the saltwater layer is observed below the freshwater layer, and the slope of the temperature and density gradients is typically steepest at the halocline. While this relationship is a key player in what shapes the water quality conditions in the estuary, there are other influences at work in the estuary as well, including wind mixing, river inflow, tidal influence, shape and size of the river mouth, air temperatures, and others.

Unfortunately, Water Agency staff were unable to implement the lagoon outlet channel during Estuary closure although three closures occurred during the management period. Consequently, there was no opportunity for Agency staff to compare the availability of suitable aquatic habitat for rearing salmonids in closed versus open Estuary conditions. However, staff were still able to collect data that provides a fuller understanding of salinity migration in the Upper Reach of the Estuary.

As freshwater flows in the Russian River decrease through spring, the salt layer typically migrates upstream. Due to continued drought conditions in the winter and spring of 2015, mainstem Russian River flows decreased earlier in the season than in 2011 and 2012, but were similar in timing to 2013 and 2014. 2015 mainstem flows were observed to drop below 200 cfs by early May. Although salinity migration patterns at the Freezeout Creek station were fairly similar to those prior monitoring years, the Brown’s Pool (RK 11.3) station had significantly less brackish water in 2015 than was observed in 2014 (Martini-Lamb and Manning 2015). Whereas the bottom of Brown’s Pool became predominantly brackish during open and closed conditions throughout the 2014 monitoring season, the bottom was only periodically brackish during open conditions in 2015. Concentrations in 2014 were as high as 10 ppt during open conditions compared to a maximum of approximately 4 ppt in 2015. Similar to 2015, Brown’s Pool remained predominantly fresh in 2013, with brief periods of brackish conditions during estuary closures in October and December (Martini-Lamb and Manning 2014). Brackish water had not been observed at Brown’s Pool prior to the 2013 monitoring season, however Water Agency staff had only previously deployed a continuously monitoring sonde at this station in the 2011 season (Manning and Martini-Lamb 2012). Even so, it is not unreasonable to expect salinity migration to periodically occur in this area, given the proximity of the Brown’s Pool station to Moscow Road Bridge (RK 10.15), where brackish water has been observed to occur.

By contrast, monitoring conducted at the bottom of the Patterson Point station in Villa Grande did not detect any significant salinity migration into the site during open or closed conditions. Maximum salinity values observed at Patterson Point were approximately 0.4 ppt, and occurred during open conditions on 25 June with flows of approximately 92 cfs. Water is considered fresh at approximately 0.5 ppt. These results correspond with the vertical profiling data collected during January 2014 in the Upper Reach of the Estuary and the MBA and further supports the theory that Brown's Riffle and the confluence of Austin Creek provide a significant hydrologic barrier to salinity migration in the mainstem Russian River.

During prolonged barrier beach closures in 2015, overall water quality conditions were observed to be similar to those of previous years. Typically during a closure or perched event, the mid-depth sondes at the Mouth, and to a lesser extent Patty's Rock and Sheephouse Creek, experience a decrease in salinity and an increase in temperature. Conversely, during prolonged closures or perched events, the upper reach of the Estuary at Freezeout Creek and Brown's Pool typically experience increases in salinity as brackish water migrates into the area, coupled with temperature increases. Conditions observed in the saline layer during the 2015 monitoring season were no exception.

DO response to Estuary closure events was variable in the Upper Reach and dependent on the presence and movement of salinity, the relative strength of stratification, circulation patterns, and flows in the Russian River. The presence of salinity would typically coincide with the presence of depressed DO levels, but not always (i.e. Freezeout Creek at the mid-depth sonde during the late November closure), suggesting that variability is dependent on relative DO concentrations in the migrating salt wedge, the length of time of Estuary closures, the timing of subsequent closure events, freshwater inflow rates, the DO concentration of inflowing freshwater, and subsequent tidal inundation and mixing.

Temperature, pH, and dissolved oxygen patterns during the 2015 monitoring season were also similar to those observed in previous monitoring years. While the Russian River Estuary is a dynamic estuarine system, the seasonal changes during the monitoring seasons have largely followed similar patterns each year since the implementation of the Biological Opinion in 2009.

To further illustrate the extent of salinity migration, a graphical representation of the maximum salinity levels recorded at various stations in the Russian River Estuary between 2009 and 2015 is being presented (Figure 4.1.47). The sondes chosen for this graph were situated in the lower portion of the water column at each station, where saline water would be expected to occur. This corresponds to approximately three to four meter depths for the Mouth, Patty's Rock, and Sheephouse Creek stations, six to nine meter depths at the Heron Rookery station, six to seven meter depths at the Freezeout Creek station, eight to ten meter depths at the Brown's Pool station, six to eight meter depths at Villa Grande, nine to eleven meters depth at Patterson Point, and one to two meters at the Monte Rio station. In the upper reaches of the Estuary and MBA, the sondes are located on the bottom of the river because the salt layer is typically thin when it occurs at these river locations. Excluding the depth variations, the graph depicts the decrease in salinity the further upstream in the Estuary and MBA the monitoring station is located.

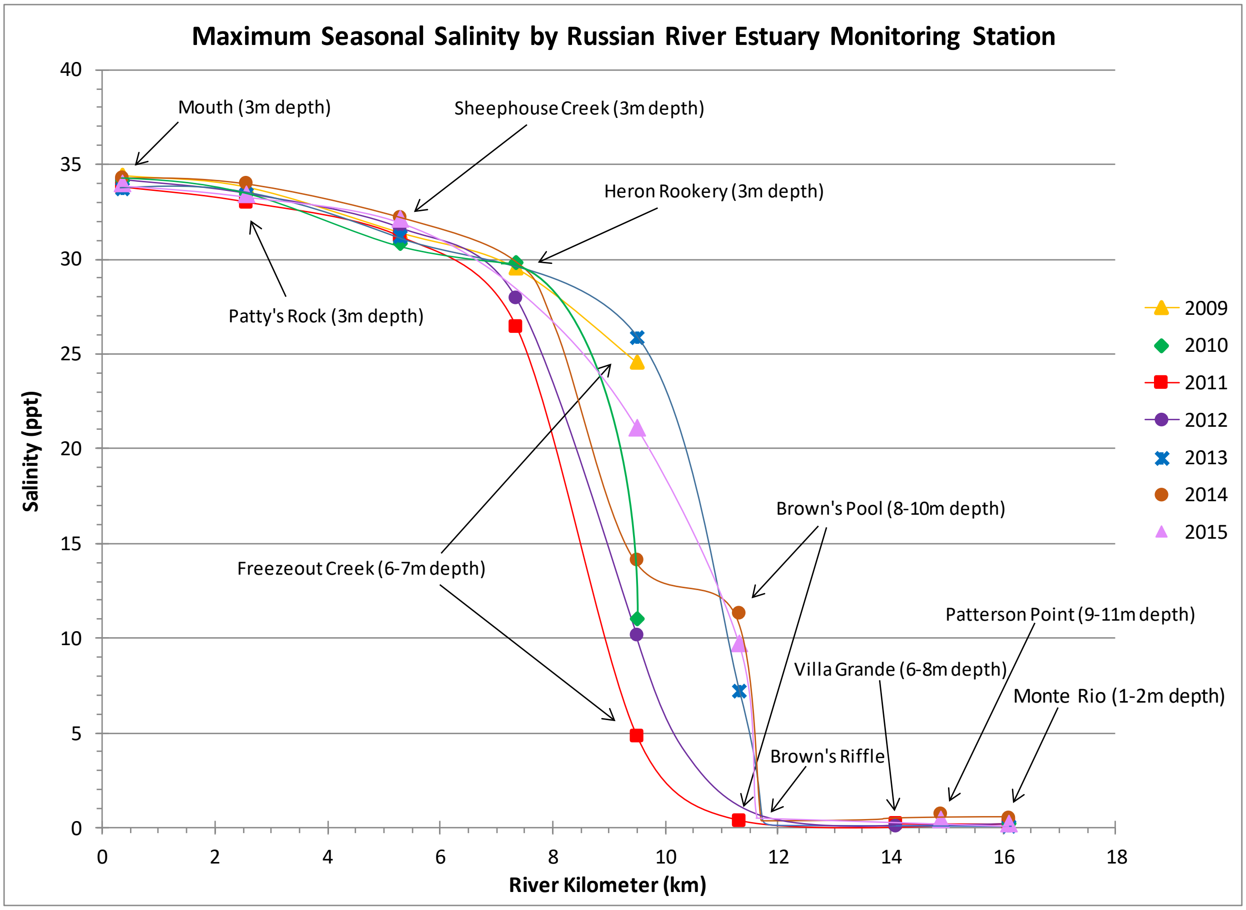


Figure 4.1.47. The maximum salinities at monitoring stations throughout the Russian River Estuary and Maximum Backwater Area between the years of 2009 and 2015.

The graph also illustrates the variable nature of salinity levels in the Upper Estuary, and specifically, one can see that the Brown's Pool maximum concentration was higher in 2015 compared to 2010 and 2013, but was lower than conditions observed in 2014 (keep in mind that the values in the graph are maximums and not means; mean values would not as clearly illustrate the degree to which brackish water was observed at Brown's Pool in 2015). Note, however, that a continuously monitoring sonde had only previously been deployed at the Brown's Pool station in the 2011, 2013, and 2014 monitoring seasons and further continuous monitoring would be necessary to determine if this degree of brackish water in the Brown's Pool is a common phenomenon.

Also note that there are no elevated salinity levels recorded at Monte Rio for any monitoring seasons. As was mentioned above, it is possible that saline water does not migrate past the riffle between Brown's Pool and the confluence of Austin Creek due to hydrologic and/or geologic conditions that serve to define a transition from the Russian River Estuary and the beginning of the Maximum Backwater Area.

Water Quality Grab Sampling Conclusions

The 2015 grab sampling effort in the Russian River Estuary continued to collect a robust set of data similar in effort to the 2012 through 2014 monitoring seasons. The increased sampling was focused on Estuary closure events and community events where water contact recreation (REC-1) was likely. Table 4.1.7 shows the total yearly number of sampling trips and the total number of samples collected within the Russian River Estuary and MBA during each monitoring season since the implementation of the Biological Opinion in 2009. There was a range of sampling events in 2015, with additional bacterial monitoring being conducted at Monte Rio and Vacation Beach in response to elevated E. coli levels at Monte Rio during summer dam removal. Although the Jenner station is located in an estuarine environment and the USEPA criteria for nutrients, chlorophyll a, and E. coli are only applicable to freshwater systems, the Jenner data is included in the calculated exceedances for year to year comparative purposes.

Table 4.1.7. The total number of grab sampling trips per monitoring season and the total number of samples taken in the Russian River Estuary and Maximum Backwater Area per monitoring season. Note; duplicate and triplicate samples were counted as separate samples during the same sampling trip.

Estuary Monitoring Season	Total Number of Sampling Trips	Total Number of Samples
2009	7	21
2010	13	65
2011	13	78
2012	18	126
2013	33	164
2014	26-31	137
2015	26-27	132

The 2015 grab sampling effort observed Total Phosphorus exceedances in 88.5% of all samples collected (Table 4.1.8). This is not uncommon in the Russian River Estuary, and similar percentages of the samples analyzed for Total Phosphorus were in exceedance during previous monitoring seasons. Table 4.1.8 shows the percentage of samples that were in exceedance each season since 2009.

The Total Nitrogen and chlorophyll a exceedances for samples taken during 2015 were also similar to percentages observed in previous monitoring years, with Total Nitrogen exceedances being lower than all previous years (Table 4.1.8). Year to year variability in the percentage of exceedances for these three constituents can be attributed in part to: the frequency and timing of storm events, fluctuating freshwater inflow rates, the frequency and timing of barrier beach closures, the strength of tidal cycles, summer dam removal, topography, relative location within the Estuary, and wind mixing.

Table 4.1.8. The percentages of samples taken that were in exceedance of U.S. EPA water quality criteria for Total Phosphorus, Total Nitrogen, and Chlorophyll a. Note; Chlorophyll a was not quantified below 0.01 mg/L in 2009, and as such, cannot be verified against the U.S. EPA criteria of 0.00178 mg/L. Also, the Total Nitrogen values in 2009 were not quantified sufficiently against the criteria to make comparisons. The U.S. EPA criteria for Total Nitrogen is 0.38 mg/L, and the criteria for Total Phosphorus is 0.02188 mg/L.

Estuary Monitoring Season	Percentage of Total Phosphorus Samples in Exceedance	Percentage of Total Nitrogen Samples in Exceedance	Percentage of Total *Chlorophyll a* Samples in Exceedance
2009	91	N/A	N/A
2010	88	23	22
2011	94	45	35
2012	73	20	16
2013	99	23	59
2014	100	14	34
2015	89	13	38

The E. coli exceedances since the implementation of the Biological Opinion in 2009 until 2015 can be seen in Table 4.1.9. However, E. coli was not sampled in 2010, with sampling being conducted for fecal coliforms instead. Although the 2014 Jenner results were not included in the calculations due to salinity conditions, the percentages of exceeded samples are still similar among sampling seasons. As was mentioned in the results section above, although Jenner is located in an estuarine environment, the 2015 data that has been determined to be reliable is included in the calculated exceedances for year to year comparative purposes. Samples collected in 2009 were analyzed using the multiple tube fermentation technique, whereas samples collected from 2011 through 2015 were analyzed using the Colilert Quanti-Tray method. Percentages for total coliform samples are not shown here since values were not quantified above 1600 MPN for 2010 and a portion of 2011, or above >2419.6 MPN for 2012, 2013 and a portion of the 2014 season. Both levels are below CDPH Guidelines, therefore it is impossible to establish percent criteria exceedances in this case.

Data collected through the grab sampling effort in 2015 appear consistent with data collected between 2009 and 2014. Further analysis could elucidate any trends that may exist temporally or longitudinally through the Russian River Estuary and guide water quality monitoring efforts in the future.

Time series trend analyses of the grab sampling data collected under the Biological Opinion could prove useful in the future. Trend analyses could determine if there have been changes over time for any of the constituents collected under this project. Certain trend tests are used for

Table 4.1.9. The percentages of samples taken that were in exceedance of CDPH Guidelines for E. coli for the sampling years 2009 through 2015. Note that for 2009, the analyzing method was multiple tube fermentation, and for 2011-2015 the method was Colilert Quanti-Tray.

Estuary Monitoring Season	Percentage of Total E. coli Samples in Exceedance
2009	5
2010	14
2011	4
2012	1
2013	3
2014	6
2015	3

non-parametric data analysis such as water quality data, including the Sen Slope test, the Kendall-Theil test, the Seasonal Kendall test, or a variety of other suitable statistical tests. Analyses of this nature require both time and expert knowledge of environmental statistical analysis. As such, they are difficult to run and outside the scope of this project at this time. In the future, allocating resources to analyses of this nature, on these data, would likely give a better understanding of the existence, or absence, of trends in the data.

4.2 Algae Sampling

Introduction

Algae sampling was conducted in the Russian River Estuary, between Patterson Point and Vacation Beach. Water Agency staff implemented the field-based rapid periphyton sampling procedure described below. Baseline conditions were sampled on September 3, 2015, and follow up sampling was conducted at every 2 foot rise in water surface elevation following closure of the estuary (sample dates of September 21 and October 1).

Methods

Periphytic Algae and Cyanobacteria

Monitoring for presence of periphytic algae in newly flooded shoreline areas was conducted during river mouth closures from May 15 to October15. Transects to monitor periphytic algal growth, including the potential presence of cyanobacteria, were established at the 3 surface water sites located in the maximum backwater area (Figure 4.2.1). Sampling was conducted along shallow over-bank habitat that becomes inundated during river mouth closure and may provide additional habitat substrate for algal mats to grow.

Transects were located on gravel bars that become inundated during estuary closure on the downstream side of Patterson Point beach, in the vicinity of the island downstream of Monte Rio, and on the gravel bar downstream from the Vacation Beach summer dam. Sampling methodology was developed based on modification of *Standard Operation Procedures for Collecting Stream Algae Samples and Associated Physical Habitat and Chemical Data for Ambient Assessments in California* (Fetscher et al. 2009) to address monitoring periphytic algae growth in newly flooded shoreline areas.

Two transects were established at each monitoring site. Transect endpoint 0 was established at a 1 m depth in the main stem Russian River and extended 12.5 m landward or to a 9 foot elevation as diagramed in Figure 4.2.2. Transect locations avoided locations such as tributaries, outfalls, and man-made structures to minimize influence of algal growth from contributions in nutrients, temperature, or canopy cover from such sources.

Percent algal cover was calculated as an algal indicator of productivity measured as algal abundance using a point-intercept collection methodology. Algal cover is the amount of microalgae coating and macroalgae taken at 5 equidistant points along each transect. The percentage of the points across the transects at each monitoring site then provide an estimate of percent algal cover.

The presence of algae was recorded for each point along the transect and identified as microalgae or macroalgae. Microalgae is defined as a “film-like coating” of algae. Measurement of microalgae thickness followed the method identified in Fetscher, et al. 2009 and an estimate of film-like coating followed descriptions in Table 4.2.1. Thicker microalgae layers were

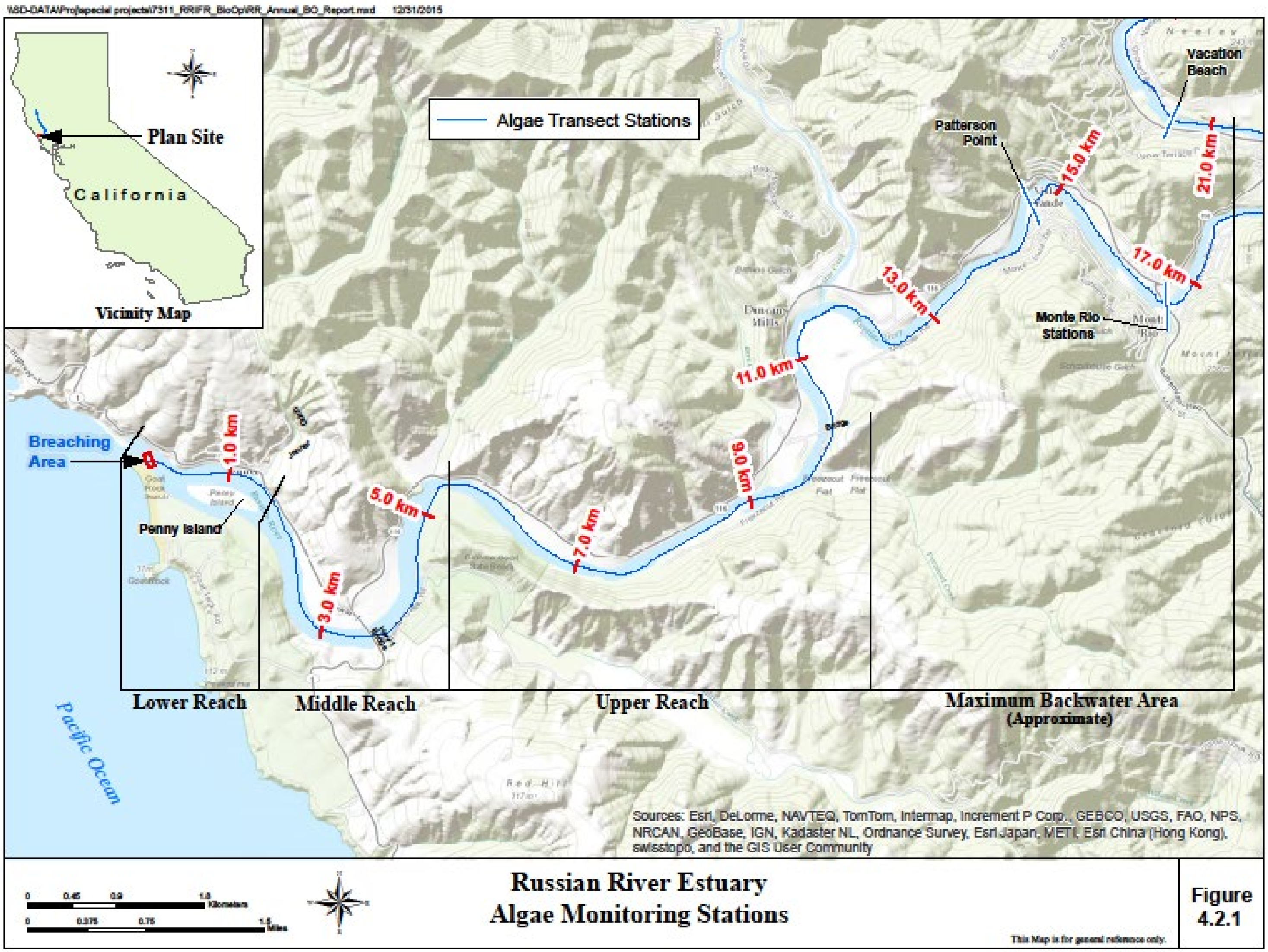


Figure 4.2.1. Algal monitoring transect locations.

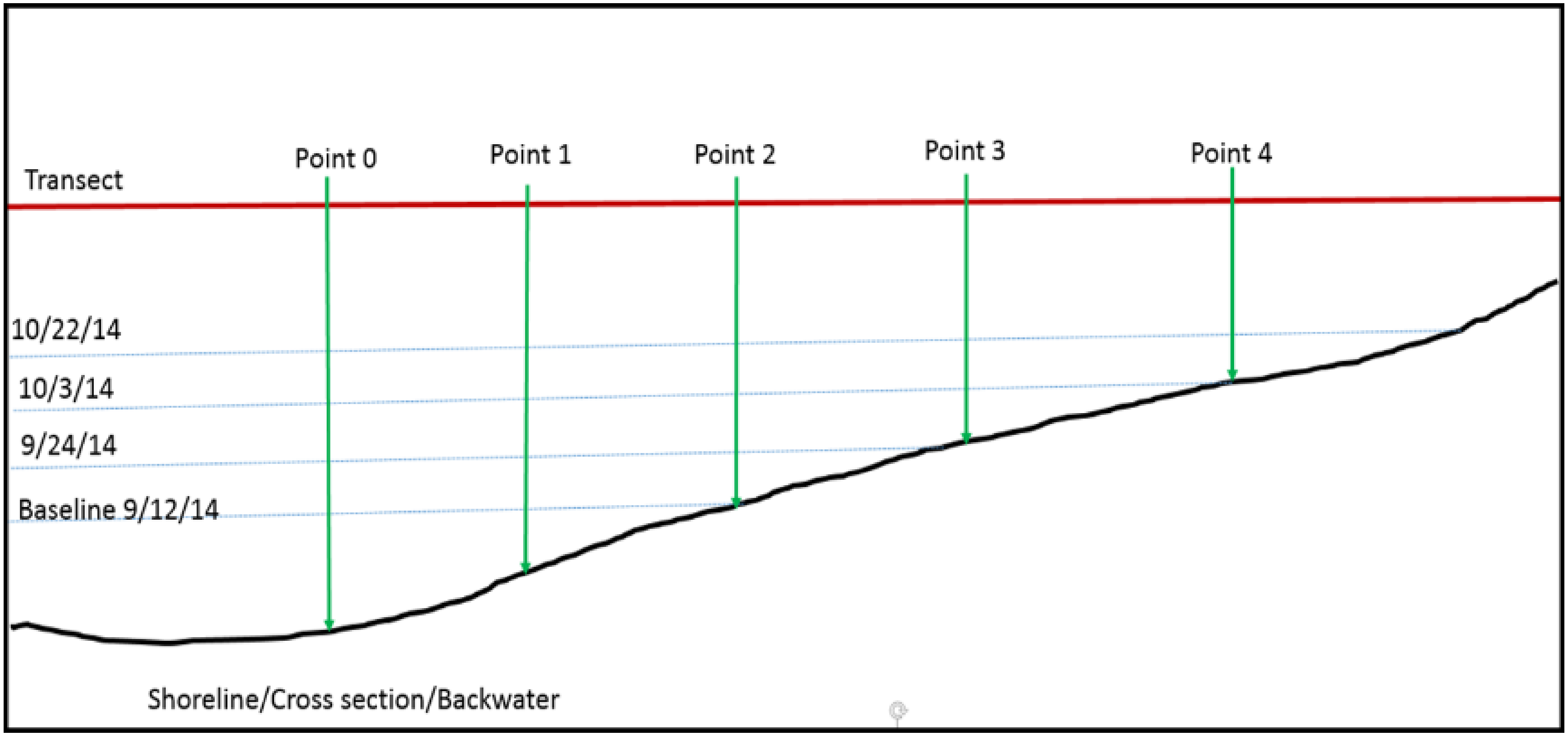


Figure 4.2.2. Transect schematic indicating transect sampling points and a representation of water levels following closure.

Table 4.2.1. Microalgal thickness codes and descriptions (from Fetscher, et al. 2009 and adapted from Stevenson and Rollins 2006).

Code	Thickness	Diagnostics
0	No microalgae present	The surface of the substrate feels rough, not slimy.
1	Present, but not visible	The surface of the substrate feels slimy, but the microalgal layers is too thin to be visible.
2	<1mm	Rubbing fingers on the substrate surface produces a brownish tint on them, and scraping the substrate leaves a visible trail, but the microalgal layers is too thin to measure.
3	1-5mm	
4	5-20mm	
5	>20mm	
UD	Cannot determine if a microalgal layer is present	

measured using a ruler or rod with demarcations at 1, 5, and 20 mm. The presence or absence of attached macroalgae or unattached, floating macroalgae was recorded at each point.

Prior to collection of percent algae cover, algae samples were collected 1 m downstream and adjacent to each point (to avoid trampling on samples during collection of percent algal cover data), beginning at the downstream transect. A single sample (10 cm diameter) were collected at each of the 5 equidistant points along the transect. Each sample was collected from the substrate that was uppermost within the stream and had highest possibility of sun exposure (i.e. if a thick layer of macroalgae covers the substrate, collection will include the layer). Samples were placed in a cooler to protect the algae from heat and desiccation and to preserve specimen integrity. Algal species present were identified to the lowest taxa, preferably species but at least genera. In addition, an evaluation for the presence of cyanobacteria within the algal samples was conducted. Keenan Foster, a taxonomic botanist and Principal Environmental Specialist with the Water Agency, conducted the algae identification and evaluation for the presence of cyanobacteria.

Water chemistry measurements was recorded near the substrate at each transect point using a YSI 6600 datasonde and YSI 650MDS datalogger. Conditions measured included water temperature, dissolved oxygen, specific conductance, pH, and turbidity. Water depth was taken using a stadia rod or similar device.

Monitoring and sample collection occurred under certain conditions and followed specific river management and operational events, noted below, at the sites described above.

- Transects were established during open river mouth conditions. Monitoring of percent algae cover and collection of samples was completed with establishment of the transects.
- The next monitoring and sampling event occurred when the river mouth was closed, in an extended perched condition, or with an outlet channel in place and the water surface elevation at the Jenner gage is at or approaching 4.5 feet. Monitoring and sample events were then repeated with each 2 foot stage change (e.g. 6.5 feet and 8.5 feet) until the river mouth returns to an open condition or at the end of the monitoring period (15 October).

Results

Monitoring locations were established at three sites that supported backwater habitats targeted for sampling. These locations are indicated in Figure 4.2.1 and include Vacation Beach, Monte Rio, and Patterson Point. Transects were established perpendicular to the shoreline in locations that were expected to be submerged during mouth closure. Transect endpoints were installed and initial data was collected while the river mouth was open on September 3, 2015. Following closure of the estuary on September 8, follow-up sampling was conducted on September 21, and October 1, which corresponded to an approximate water surface elevation gain of 2 feet additively for each sampling event.

Table 4.2.2 summarizes micro versus macro algal cover data. Table 4.2.3 indicates the genera encountered during sampling and notes the relative abundance during surveys. Blue green algae cover was sampled as a total estimate along with other forms of microalgae including microscopic Green Algae (Chlorophyta) and Golden Brown Algae (Chrysophyta - diatoms). Figures 4.2.3-4.2.7 illustrate the relationship and shift in relative cover by micro and macroalgae following estuary closure. Figure 4.2.3 illustrates this relationship graphically, first all sites represented in one graph together, then individually by sampling location (Figures 4.2.4-4.2.6), and finally represented as average change in cover by micro and macroalgae for all sites (Figure 4.2.7). Figures 4.2.8 through 4.2.13 site conditions at Patterson Point, Monte Rio and Vacation Beach on 21 September 2015. Figures 4.2.14 and 4.2.15 illustrate benthic drift conditions typical of back water and shoreline areas. Figures 4.2.16 through 4.2.17 provide the macroscopic view of planktonic green algae and cyanobacterial colonies typically seen in the Russian River associated with fall benthic blooms accumulating behind a barrier (flashboard dam). Figure 4.2.18 illustrates the littoral, limnetic and profundal zones in a typical freshwater system. Figure 4.2.19 shows the new waterline and freshly captured littoral zone following estuary closure. Figures 4.2.20 through 4.2.23 shows typical drift that accumulates on the shoreline and in submerged vegetation following estuary closure. Figures 4.2.24 illustrates benthic algal colonies following localized scour below Vacation Beach.

Table 4.2.2. Change in relative cover over time between micro- and macro- algae between September 3 and October 1, 2015.

Date	Sampling Location	Microalgae Cover	Macroalgae Cover	Estuary Condition
9/3/2015	Vacation Beach	53%	47%	Open (baseline)
9/21/2015	Vacation Beach	53%	47%	Closed
10/1/2015	Vacation Beach	57%	43%	Closed
9/3/2015	Monte Rio	64%	36%	Open (baseline)
9/21/2015	Monte Rio	22%	78%	Closed
10/1/2015	Monte Rio	33%	67%	Closed
9/3/2015	Patterson Point	53%	47%	Open (baseline)
9/21/2015	Patterson Point	29%	71%	Closed
10/1/2015	Patterson Point	37%	63%	Closed

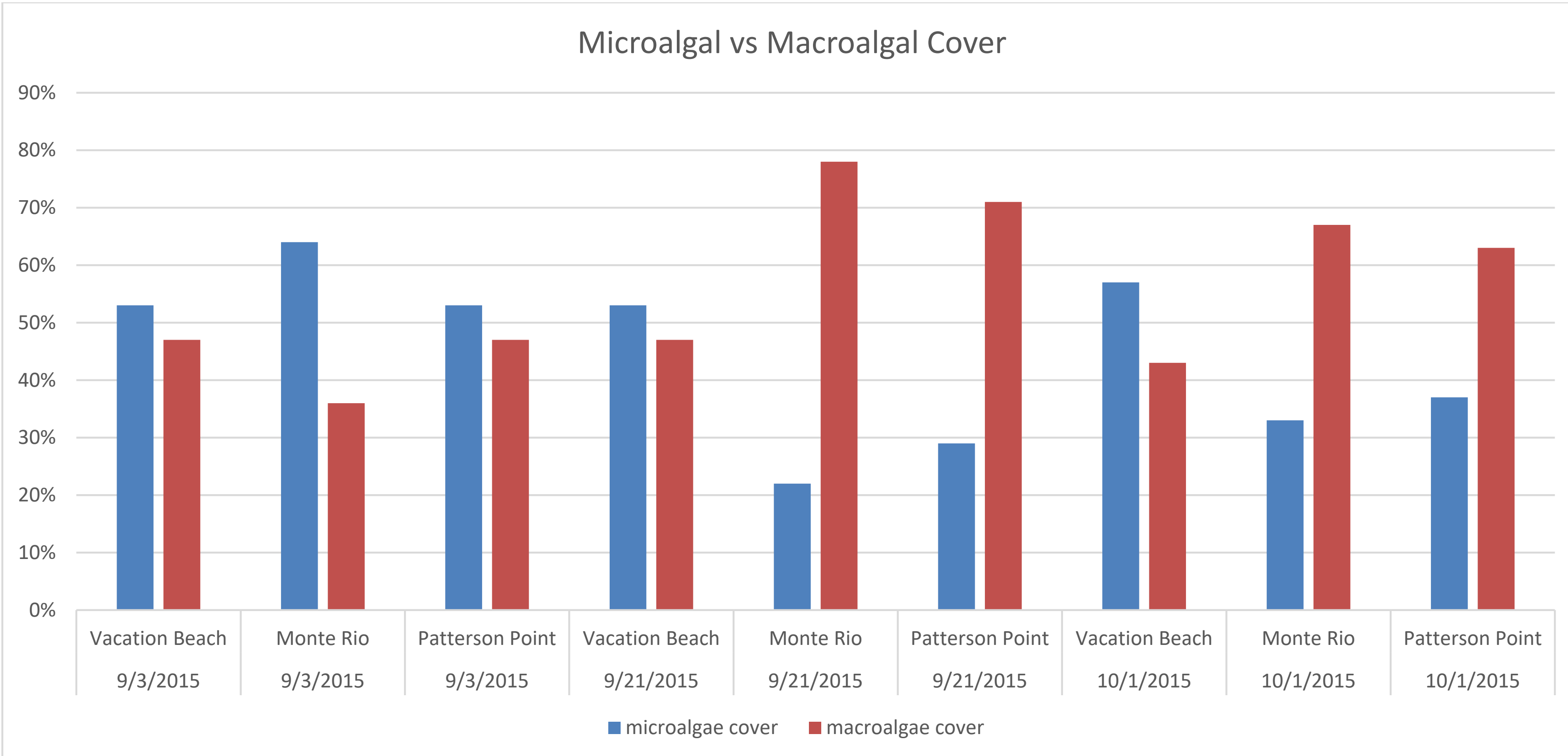


Figure 4.2.3. Change in microalgae versus macroalgae cover at all sampling sites during 2015 Russian River Estuary closure.

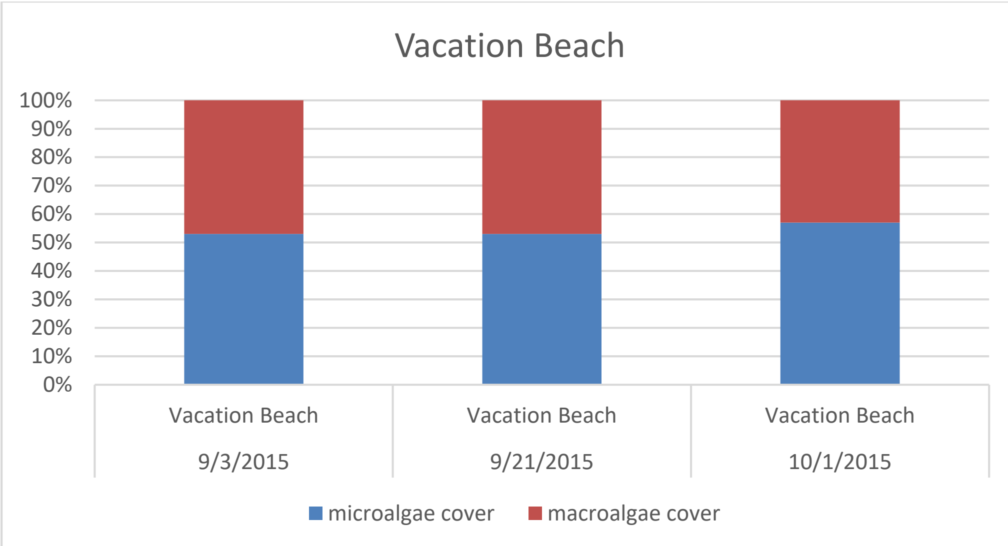


Figure 4.2.4. Change in microalgae versus macroalgae cover at Vacation Beach during 2015 Russian River Estuary closure.

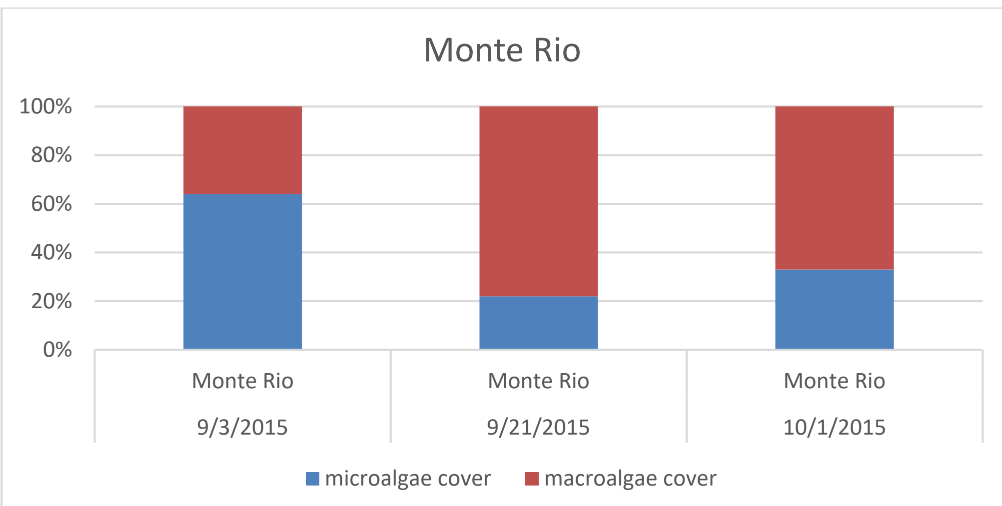


Figure 4.2.5. Change in microalgae versus macroalgae cover at Monte Rio during 2015 Russian River Estuary closure.

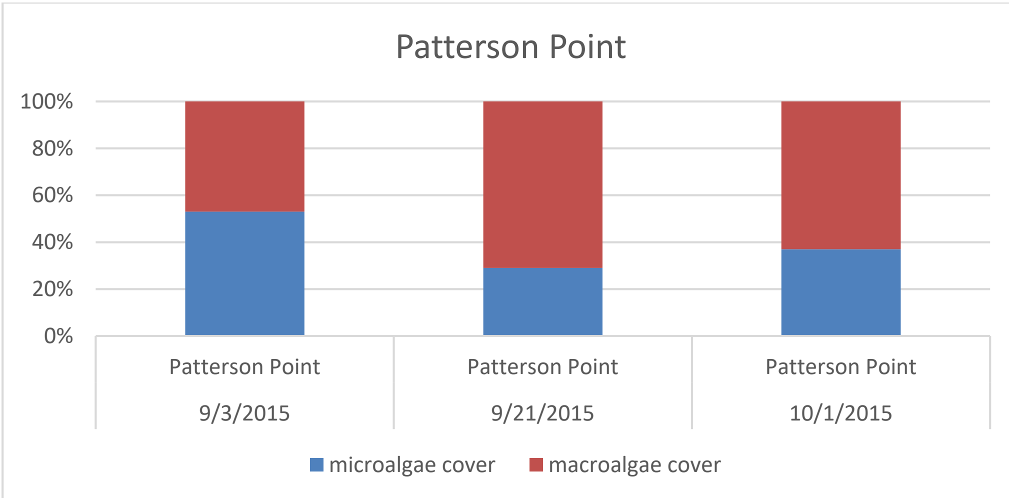


Figure 4.2.6. Change in microalgae versus macroalgae cover at Patterson Point during 2015 Russian River Estuary closure.

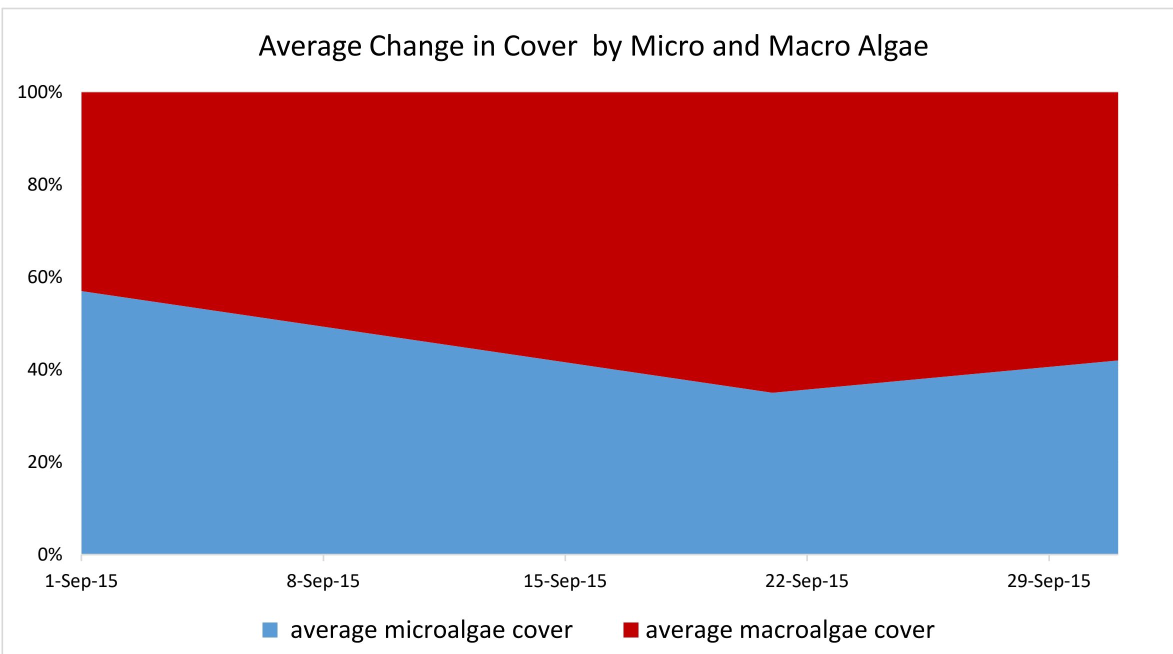


Figure 4.2.7. Average change (at all sites) in microalgae versus macroalgae cover during 2015 Russian River Estuary closure.

Figure 4.2.8 and Figure 4.2.9. Sampling at Patterson Point. September 21, 2015.

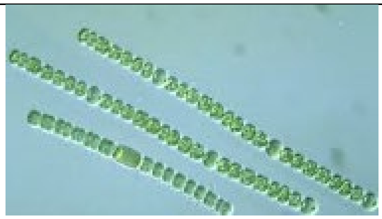
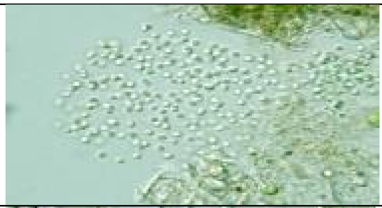
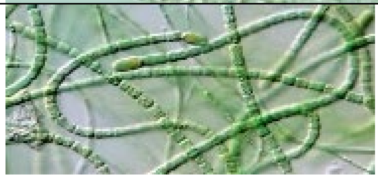
Figure 4.2.10 and Figure 4.2.11. Sampling at Monte Rio. September 21, 2015.

Figure 4.2.12 and Figure 4.2.13. Sampling at Vacation Beach. September 21, 2015.

Figure 4.2.14 and Figure 4.2.15. Typical planktonic periphyton in the Lower Russian River. Complex mixture of filamentous green algae (Cladophora, Zygnema, Spirogyra) (mostly lighter green), mixed diatoms (golden brown color), and mixed cyanobacterial benthic colonies (Anabaena, Cylindrospermum, Gleotricha) (pine to blue green).

Figure 4.2.16 and Figure 4.2.17. Flashboards at Vacation Beach, algal drift accumulates at the bottom end. Composed of various detritus and mixed colonies of filamentous green and colonial blue green algae species.

Table 4.2.3. Genera Observed during Algal Monitoring September – October 2015

Algal Class	Genus	Notes on Occurrence/Ecology[2]	Known Toxins	Photograph
Cyanophyta	Anabaena	Common: individual or in colonies masses of individual filaments no sheath, common, easy to confuse with Nostoc sp. if gelatinous sheath indiscernible. Saxicolous, goes planktonic later in season, possibly stimulated by estuary closure, or shortening day, accumulates on shoreline in backwater areas.	Microcystins, Anatoxin, Saxitoxins	
Cyanophyta	Ahanocapsa sp.	Occasional: colonies embedded in detritus on fine substrate.		
Cyanophyta	Cylindrospermum sp.	Common: saxicolous, goes planktonic later in season, possibly stimulated by estuary closure, or shortening day, accumulates on shoreline in backwater areas	Anatoxin	
Cyanophyta	Gloeotricha sp.	Occasional: forms brownish hollow, gelatinous thallus. Saxicolous then planktonic, accumulates on shoreline in backwater areas.		

[2] Note- Common- Observed in 90% of samples, Occasional –Observed in about 50% of samples, Rare-Observed in only one sample.

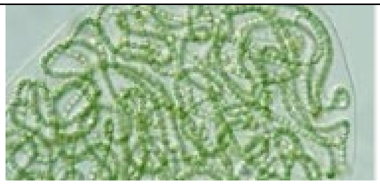
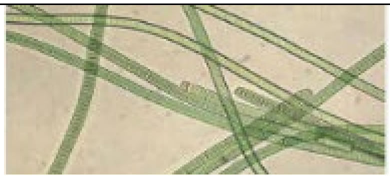
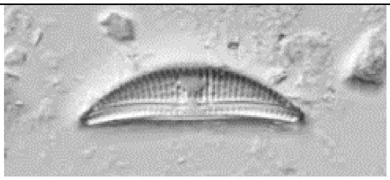
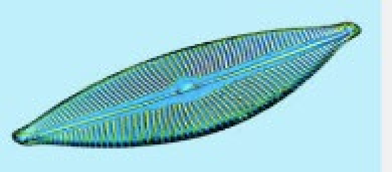
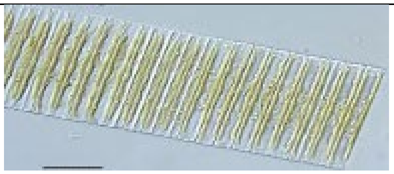
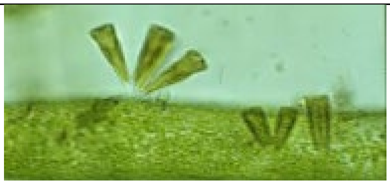
Algal Class	Genus	Notes on Occurrence/Ecology[2]	Known Toxins	Photograph
Cyanophyta	Nostoc sp.	Occasional: forms small gelatinous hollow balls.	Microcystins	

Table 4.2.3 (cont.). Genera Observed during Algal Monitoring September - October 2015

Algal Class	Genus	Notes on Occurrence/Ecology	Known Toxins	Photograph
Cyanophyta	Oscillatoria sp.	Common: forms flat threadlike colonies, very dark blue green in color, or occurs individually.	Microcystins	
Bacillariophyta	*Amphora* sp.	Common in diatomaceous layer on substrate		
Bacillariophyta	*Cymbella* sp.	Common in diatomaceous layer on substrate		
Bacillariophyta	*Fragilaria* sp.	Common in diatomaceous layer on substrate		
Bacillariophyta	*Gomphonema* sp.	Common in diatomaceous layer on substrate and debris. Most abundant species observed. Golden brownish in color, epiphyte on macroalgae		

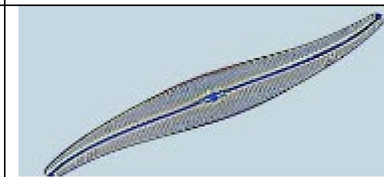
Algal Class	Genus	Notes on Occurrence/Ecology	Known Toxins	Photograph
Bacillariophyta	*Gyrosigma* sp.	Common in diatomaceous layer on substrate		

Table 4.2.3 (cont.). Genera Observed during Algal Monitoring September - October 2016

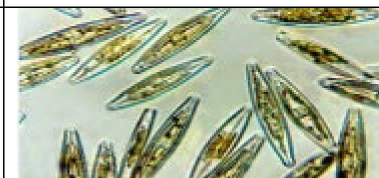
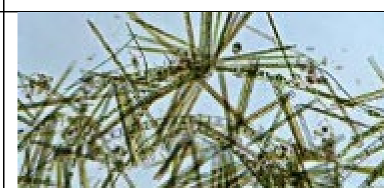
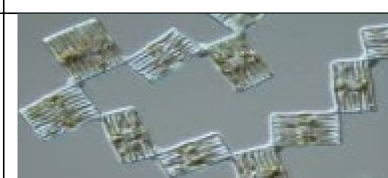
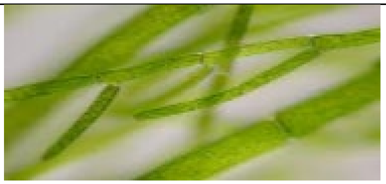
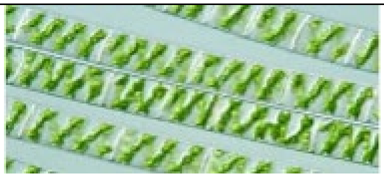
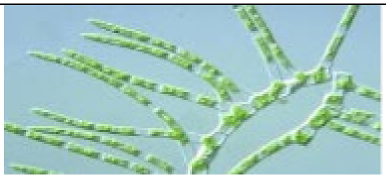
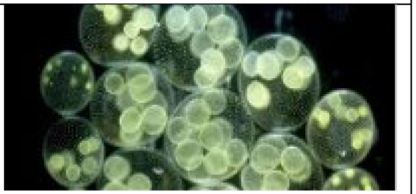
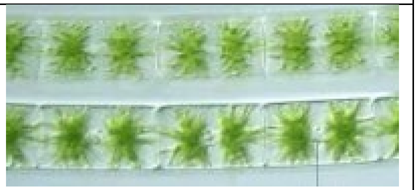
Algal Class	Genus	Notes on Occurrence/Ecology	Known Toxins	Photograph
Bacillariophyta	*Melosira* sp.	Generally marine species, likely carried in with the tide.		
Bacillariophyta	*Navicula* sp.	Common in diatomaceous layer on substrate.		
Bacillariophyta	*Surirella* sp.	Common in diatomaceous layer on substrate.		
Bacillariophyta	*Synedra* sp.	Common in diatomaceous layer on substrate.		
Bacillariophyta	Tabellaria sp.	Common in diatomaceous layer on substrate.		

Table 4.2.3 (cont.). Genera Observed during Algal Monitoring September - October 2015

Algal Class	Genus	Notes on Occurrence/Ecology	Known Toxins	Photograph
Chlorophyta	Cladophora sp. (few species)	Very Common: dark pine green, saxicolus, branching filament, reticulate chloroplast multiple pyrendoids. Dominant benthic green, provides substrate for diatom a cyanobacteria colonies. Goes planktonic when reproductive.		
Chlorophyta	Mougeotia sp.	Occasional: Vacation Beach, drift at Patterson Point		
Chlorophyta	Spirogyra sp. (at least 3 diff species)	Very Common: light green, saxicolous, unbranched filament, helical chloroplast with multiple pyrenoids. Goes planktonic when reproductive. Slippery cell walls, feels slimy.		
Chlorophyta	Stigeclonium sp.	Occasional: saxicolous (vacation beach) branched bright green.		
Chlorophyta	Volvox sp.	Rare: in one sample (Patterson Point)		

Algal Class	Genus	Notes on Occurrence/Ecology	Known Toxins	Photograph
Chlorophyta	Zygnema sp.	Very Common: light green, saxicolous, unbranched filament, platelike chloroplast. Goes planktonic when reproductive. Two star shaped chloroplasts per cell		

Discussion/Observations

Algae occurs in the lower Russian River and Estuary under a variety of conditions and species commonly found worldwide are present in the system. Conditions supporting algal abundance are largely driven by light, temperature, stream flow, and nutrient availability. Generally the most visible type of algae are filamentous Green Algae (Family Chlorophyta) initially growing on rocks and substrate (generally cobble, gravels, and occasionally finer grained sands and silts) (saxicolous) and then becoming planktonic during their reproductive phase, which is driven by largely by season, unless another environmental parameter changes and triggers the life cycle switch (light, temperature, nutrient availability, and changes in water depth). Figure 4.2.18 illustrates a representative cross section of a water body, showing the littoral, limnetic, and profundal zones. Figure 4.2.19 indicates these zones at Monte Rio following a two-foot increase in water surface elevation. The profundal zone is below the area of active photosynthesis, and in the Russian River, generally in areas that exceed 3 feet in depth depending on water clarity. Depending on the annual conformation of the substrate following high flow events, the littoral zone may be larger or smaller depending on where the river moved the substrate during functional flows. Cover data from 2014 and 2015 on micro versus macroalgae indicate that following estuary closure and the following slow increase in depth (with the corresponding reduction in what used to be photosynthetically active littoral zone) there is a shift in algal dominance (cover) from micro-algae dominated to macro-algae dominated. This shift is associated with all forms of algae and is triggered by environmental change. In this case the environmental change is the increasing water depth and the corresponding shift in the base elevation of the column of water that can be penetrated by sunlight.

Green Algae

Common green algae genera in the lower Russian River and Estuary include *Chladophora* sp, *Spirogyra* sp, and *Zygnema* sp. Besides diatoms (described below), Green Algae is one of the most prevalent types of algae recognizably visible at the macro-scale. *Chladophora* is a common branching green alga (often slightly darker green) that grows on rocks and is observed in almost every habitat niche available (cobble, gravel, shallow, fast, deep, slow, shaded, direct sun, etc.) in the littoral zone. The greens and in particular (*Cladophora* sp.) appear to provide the substrate base for the periphyton (complex mixture of algae, detritus, and microbes). Early in the season the filaments are lightly colonized by diatoms and cyanobacterial colonies. Later in the season Cladophora filaments are densely encrusted with free living and tube dwelling diatoms and gelatinous cyanobacterial colonies. Flow also affects what can be retained in the periphyton. Fast water can preclude ultimate stature and size of the periphyton as velocity tends to shear off individual accumulations larger than four to six inches in length. Species diversity comparisons between samples collected in high flow areas indicate that high flows encourage filamentous and colonial forms over free living diatoms. Large substrate (submerged wood, cobble, large gravels, aquatic plants) allows filamentous greens and associated periphyton to reach their maximum sizes. In backwater areas, or locations with sluggish flow at the water edge, the *Chladophora* generally gets completely encrusted in diatoms and cyanobacteria colonies. These green algae start their growth attached to the substrate but if physically disturbed (walking, swimming, rapid flow changes) or when forming reproductive propagules (generally in the Fall) the filaments detach and form large floating and visible rafts (these can negatively affect dissolved oxygen while they are decomposing). Often the green algae or emergent plants provides a substrate for other forms of algae, including diatoms (Figure 4.2.28), unicellular greens, and cyanobacteria. Floating mats dominated by green algae were observed to include in varying proportions a wide variety of other algal genera including diatoms, cyanobacteria, and other greens.

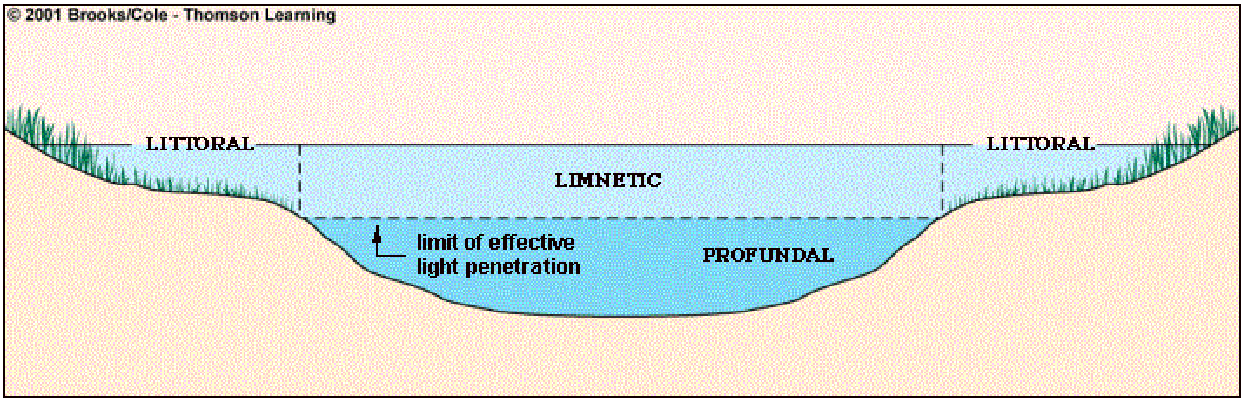


Figure 4.2.18. Diagram indicating littoral vs limnetic and profundal zones. Following river mouth closure, the profundal zone moves into the littoral zone and existing benthic algae either detach or if they have the means, move and re-colonize the newly wetted littoral zone.

Figure 4.2.19. Diagram indicating newly submerged littoral zones at Monte Rio, September 21, 2015.



Figures 4.2.20, 4.2.21, 4.2.22, and 4.2. 23. Following river mouth closure, filamentous green algae with mixed diatoms and cyanobacteria colonies re-colonizes the newly wetted littoral zone by accumulating at water's edge, drifting into backwaters and getting entrained by vegetation that grows on the submerging gravel bars.

Figure 4.2.24. Benthic colonies scoured (showing zonation patterns) following flashboards being taken down on Vacation Beach. Base of colonies dominated by cyanobacterial colonies (Anabaena, Cylindrospermum, and Gleocapsa).

Golden Brown Algae

The most numerous and abundant type of algae found in most freshwater systems, and true for the Russian River as well, are diatoms, members of the Golden Brown Algae (Family Chrysophyta). These algae develop siliceous (glass) cell walls called "frustules" and display a wide range of shapes and sizes. Diatoms comprise the majority of the micro-algal crusts and fluffy brown growths found on submerged substrate in the photic zone (littoral) (Figure 4.2.18 and 4.2.19). Diatoms have a variety of life styles and can be found as free-swimming (gliding) individuals, colonies of hundreds to thousands cells that form and live together in gelatinous tubes, and in long filaments (Figure 4.2.39). They make up a large part of the periphyton and were commonly observed mixed in the "planktonic drift" following river mouth closure. Diatoms are the first algal species along with cyanobacteria that colonize fresh substrate to form biofilms that support algal succession of the periphyton as flows reside and water levels drop in the spring (Bellinger 2015).

Cyanobacteria

Cyanobacteria or “blue green algae” are bacteria that, like plants, use solar energy and carbon dioxide to grow. As bacteria (procaryotes) they lack the complex cellular organization found in eucaryotic cells (nucleus, mitochondria, chloroplasts, endoplasmic reticulum, etc.). Cyanobacteria occur naturally in both freshwater and marine (salt) water bodies. Cyanobacteria are extremely common in the shallow water habitats along the Russian River. Dominant cyanobacterial genera sampled include Anabaena, Gleotrichia, Cylindrospermum, and Ocillatoria (Phormidium).

Toxic cyanobacteria are found worldwide in inland and coastal water environments. At least 46 species have been shown to cause toxic effects in vertebrates (WHO 2003). The most common toxic cyanobacteria in fresh water are Microcystis spp., Cylindrospermopsis raciborskii, Planktothrix (syn. Oscillatoria) rubescens, Synechococcus spp., Planktothrix (syn. Oscillatoria) agardhii, Gloeotrichia spp., Anabaena spp., Lyngbya spp., Aphanizomenon spp., Nostoc spp., some Oscillatoria spp., Schizothrix spp. and Synechocystis spp. Toxicity cannot be excluded for further species and genera (WHO 2003).

Blooms

Algae are photosynthetic microorganisms that are found in most habitats. Algae vary from small, single-celled forms to complex multi-cellular forms. An algal bloom is a rapid increase in the density of algae in an aquatic system. Algal blooms sometimes are natural phenomena, but their frequency, duration and intensity are increased by nutrient pollution. Algae can multiply quickly in waterways with an overabundance of nitrogen and phosphorus, particularly when the water is warm and the weather is calm. This proliferation causes blooms of algae that turn the water noticeably green, although other colors can occur. Some species of algae grow in clumps covered in a gelatinous coating and have the capability to float, allowing cells to stick together into large surface scums in calm weather. Other algae form thick mats that float on or just below the surface along the shoreline. In the lower Russian River, accumulations of algae floating at the surface have been observed to be composed of green algae, cyanobacteria, and diatoms. These “blooms” have been sampled and are composed of discrete aggregates of what used to be attached to the substrate as part of the periphyton (clumps of detritus mixed with whole colonies of different genera of cyanobacteria, green algal reproductive spores, partially decayed filamentous green algal genera, tube dwelling diatoms, and individual trichomes of Oscillatoria or Phormidium, etc.).

Most algae species go planktonic when entering their reproductive phase and can form large floating mats in backwater areas that locally affect dissolved oxygen as the thallus (algal body) disintegrates into propagules (resting spores, aplanospores, akinetes). Stimulus to convert algal metabolism from vegetative to reproductive is tied to light and substrate availability in conjunction with water quality, nutrient availability, and the average life cycle of the species in question. Spring through early fall are the times of year that water bodies typically exhibit the most visible response to water quality problems. Algal blooms can be dramatic and can be a result of excess nutrients from fertilizer, wastewater and storm water runoff, coinciding with lots of sunlight, warm temperatures and shallow, slow-flowing water. The challenge is separating a bloom caused through natural stimuli (reduced insolation from shorter days, increased shading due to inclination of the sun, leading to cooler water temperatures and slower metabolism) from the bloom caused from man-induced stimuli (un-natural fertilizer inputs, stirring up substrate, artificially modifying depth of littoral zone, etc.).

Rivers are not known for having cyanobacterial blooms that are composed of individual cells in the water column. Generally rivers are similar to oligotrophic lakes with low nutrient content in the water. Algal blooms in rivers are generally a result of the benthic genera (periphyton) going planktonic because of an environmental change or the end of the life cycle of a clone. These benthic mats can only grow in clear water where sunlight penetrates to the bottom, and reach their greatest development in locations with high light intensities. During sunny days, especially in the fall, photosynthesis drives oxygen production which forms bubbles in the colony mats (making up the periphyton) that loosen parts of the mats and drives discrete clumps of them to the surface. Mats and broken bits of benthic cyanobacteria colonies wash up on the shore line and can be a hazard if ingested (Figures 4.2.14, 4.2.15, 4.2.16, 4.2.17, 4.2.22, and 4.2.23). These mats may be potentially lethal to animals when ingested, depending on the species and if toxins are released. The human impact of benthic cyanobacterial mats is less than from planktonic blooms in the water column, but is worth noting as these kinds of waters, or algae in this form is not generally recognized as producing cyanotoxins (WHO 2003).

Cover Shifts

Cover data displayed in Table 4.2.2 and represented in Figures 4.2.3 through 4.2.7, are indicative of the shift in cover triggered by water level increase. Generally the data collected in 2015 is similar to results in 2014. Vacation Beach did not show a conclusive shift from microalgae to macroalgae dominance. Vacation Beach is the farthest sampling point from the mouth of the Russian River and the effects of closure (water level increase) were inconclusive during the three sampling events.

Generally data support the observation that water level rise causes the benthic mats of microalgae to detach from their locations in the littoral zone and through shoreline accumulation of floating colonies (and motile cells) begin to re-colonize the freshly wetted gravel bars, and other newly inundated low-lying areas. Figure 4.2.25 diagrammatically illustrates conditions before closure. Benthic algae is found in the photosynthetically active littoral zone but drops off in abundance quickly below the littoral zone. Figure 4.2.26 illustrates conditions following closure. In most cases, the area of habitat in the littoral zone increases as the water surface elevation increases. The benthic algae and periphyton break away from the substrate and drift onto the shoreline. Motile genera including diatoms start colonizing the new areas but where not observed re-developing into the thick crust present before estuary closure.

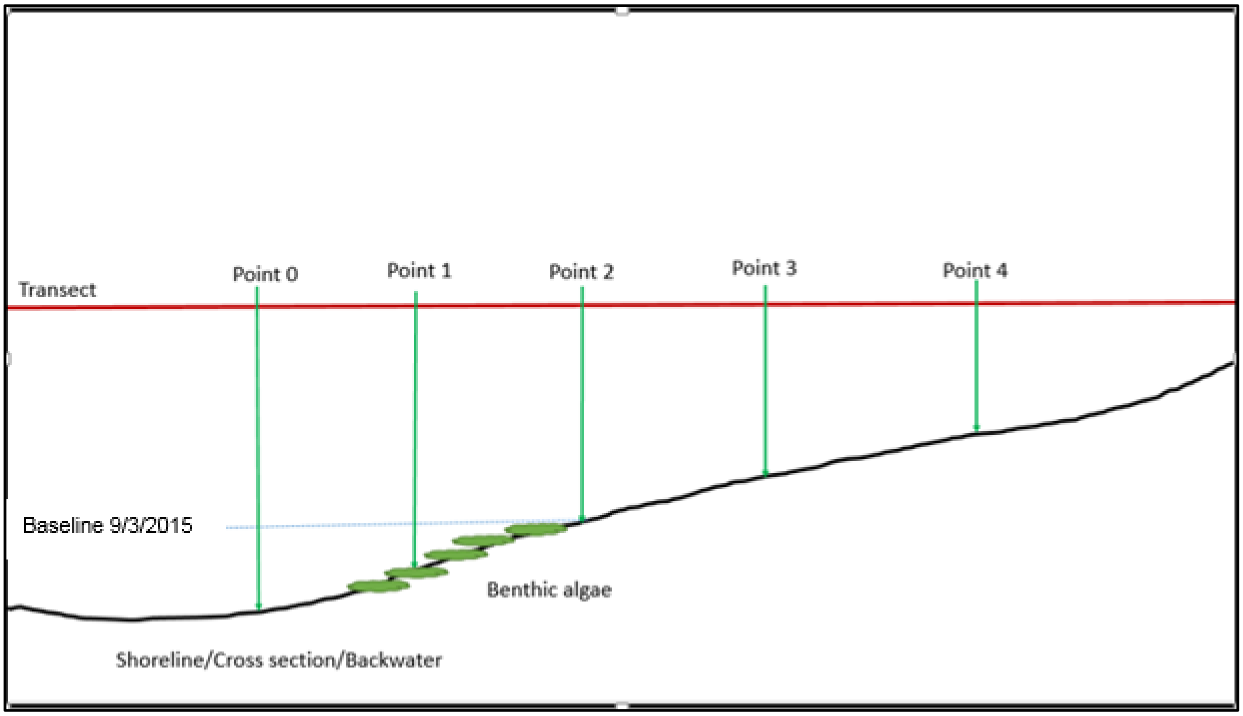


Figure 4.2.25. Before the river mouth closes algae is spread relatively evenly across the littoral zone.

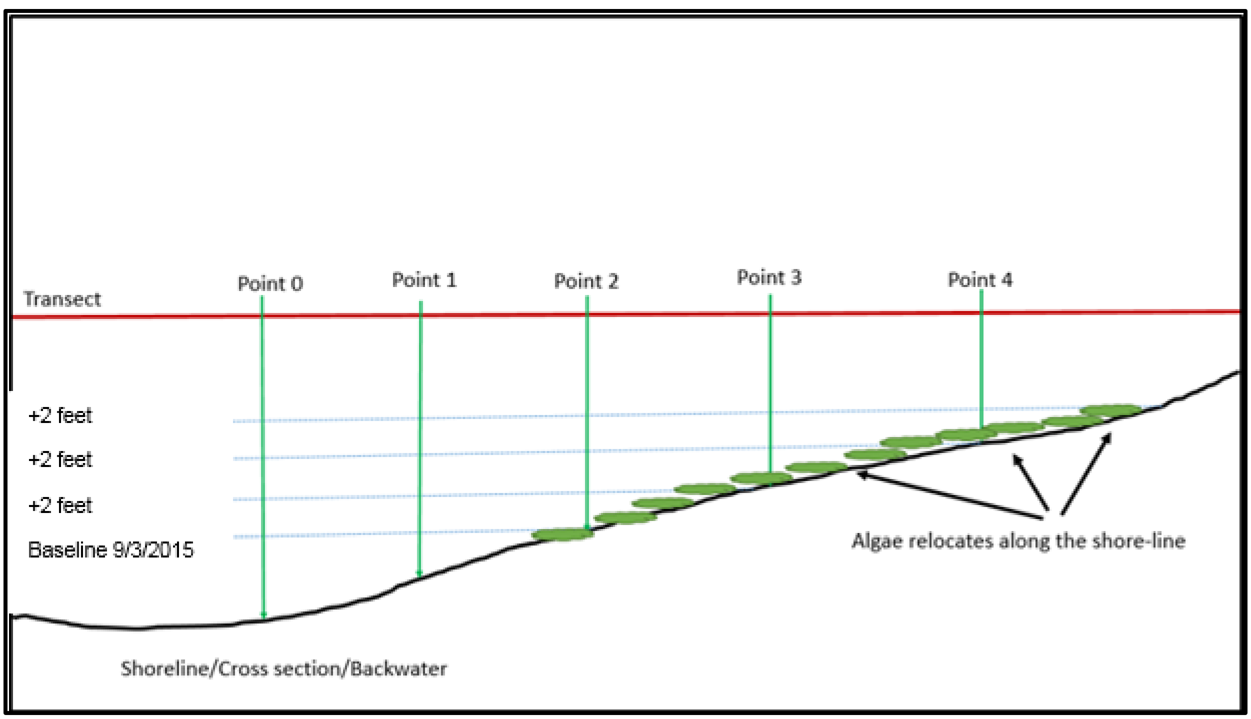


Figure 4.2.26. After the river mouth closes algae moves upslope either by drift or active motility and colonizes the newly wetted littoral zone.

Recommendations

There is a response to river mouth closure observed and measured during algae sampling/monitoring. The current methods of sampling cover does not provide data on what genera are comprising the cover, their distribution in the river, assess available habitat in the photozone, nor evaluate the physical or physiological triggers that stimulate the periphyton to turn planktonic. Further analysis would be helpful to understand the shifts in algal cover by genera over the growth season. Studying initial recolonization following spring scour through to fall reproductive blooms would be helpful to better understand both the genera and successional processes involved.

Further taxonomic work should be done to identify the cyanobacteria in the Russian River to the species level as species toxicity can vary widely across individual genera. Studies should be designed to determine under what conditions or if these colonies release or retain their cyanotoxins during planktonic periods in their life cycles. Determining what factors lead benthic cyanobacterial colonies and or “benthic blooms” to release their toxins would assist in determining hazard associated with these floating colonies. Benthic sampling should be expanded to evaluate the planktonic algae occurring in the water column so they can be evaluated specifically for their taxonomy and abundance as well.

4.3 Invertebrate Prey Monitoring, Salmonid Diet Analysis and Juvenile Steelhead Behavior

Introduction

The Russian River Biological Opinion requires the Sonoma County Water Agency (Water Agency) to “monitor the effects of alternative water level management scenarios and resulting changes in depths and water quality (primarily salinity, dissolved oxygen concentration, temperature, and pH) on the productivity of invertebrates that would likely serve as the principal forage base of juvenile salmonids in the Russian River Estuary (NMFS 2008). Specifically, Water Agency is determining the temporal and spatial distribution, composition (species richness and diversity), and relative abundance of potential prey items for juvenile salmonids in the Estuary, and evaluating invertebrate community response to changes in sandbar management strategies, inflow, estuarine water circulation patterns (stratification), and water quality. The monitoring of invertebrate productivity in the Estuary focuses primarily on epibenthic and benthic marine and aquatic arthropods within the classes Crustacea and Insecta, the primary invertebrate taxa that serve as prey for juvenile salmonids, especially steelhead (*Oncorhynhus mykiss*) that may be particularly characteristic of conditions unique to estuarine lagoons for which steelhead may be adapted in intermittent estuaries near the southern region of their distribution (Hayes and Kocik 2014). The monitoring effort will involve systematic sampling and analysis of zooplankton, epibenthic, and benthic invertebrate species” (NMFS 2008, page 254).

Commensurate with assessment of potential responses to Estuary conditions by the macroinvertebrate prey of juvenile salmonids, the Water Agency is also monitoring juvenile salmonid diet composition and behavior. Based on the hypothesis that both diet and behavior of juvenile salmonids will vary as a function of increased water level and rearing space when the mouth of the Estuary is closed, the potentially differential effects of density-dependent interactions on diet composition and consumption rate are being compared between open and closed Estuary conditions. To facilitate the synthesis of this information with more precise information on juvenile salmonid exposure to variability in Estuary salinity and thermal regime, the Water Agency is supporting hydroacoustic telemetry of their position, behavior and residence as a function of Estuary conditions. The purpose of this effort is to determine for juvenile steelhead in the Estuary between June-September the variation under different Estuary open-closure conditions in: (1) the Estuary’s water quality environment and the specific water quality conditions experienced by the juvenile steelhead; (2) their behavior in terms of estuarine habitat, reach occupancy and intra-estuarine movement patterns; (3) diet composition; (4) potential (modeled) and empirical growth. These will be used to refine parameters used in the Seghesio (2011) bioenergetics model to generate more empirically-based potential growth estimates during juvenile steelhead response to changing conditions in this intermittent Estuary.

The Water Agency entered into an agreement with the University of Washington, School of Aquatic and Fishery Sciences’ Wetland Ecosystem Team (UW-WET) to conduct studies of the ecological response of the Estuary to natural and alternative management actions associated with the opening and closure of the Estuary mouth. This component of the Biological Opinion study is designed to evaluate how different natural and managed barrier beach conditions in the Estuary affect juvenile salmon foraging and their potential prey resources over different temporal and spatial scales. Systematic sampling is intended to capture the natural ecological responses (prey composition and consumption rate) of juvenile salmon and availability of their prey resources (insect, benthic and epibenthic macroinvertebrates, zooplankton) under naturally variable, seasonal changes in water level, salinity, temperature and dissolved oxygen conditions. A second approach, event sampling, was originally proposed in 2009 to contrast juvenile salmonid foraging and prey availability changes over Estuary closure and re-opening events. The hydroacoustic telemetry component was particularly adaptable and targeted for the event sampling.

Methods

Sampling Sites

Sampling for fish diet and prey availability is designed to coincide with established Water Agency and other related sampling sites distributed in the lower, middle, and upper reaches of the Estuary during the Lagoon Management Period (May 15 to October 15). Since 2009, salmonid diet samples have been coincident with beach seining at 11 primary sites (Figure 4.3.1; modified from Largier and Behrens 2010) sampled for juvenile salmon by the Water Agency – (1) Lower Reach: River Mouth, Penny's Point and Jenner Gulch; (2) Middle Reach: Patty's Rock, Bridgehaven and Willow Creek; and, (3) Upper Reach: Sheephouse Creek, Heron Rookery, Freezeout Bar, Moscow Bridge and Casini Ranch. When possible, samples are specifically selected for diet analysis from the overall beach seine collections at Jenner Gulch to represent the lower Estuary reach, Bridgehaven to represent the middle reach and Casini Ranch, Freezeout Bar and Sheephouse Creek to represent the upper reach. Incidental steelhead diet samples also originated from Penny Point (lower), Willow Creek (middle), and Casini Ranch (upper) sites when there are not sufficient samples from the preferred reach sites. These locations also overlap with sites established by water quality measurements—dissolved oxygen, temperature and salinity.

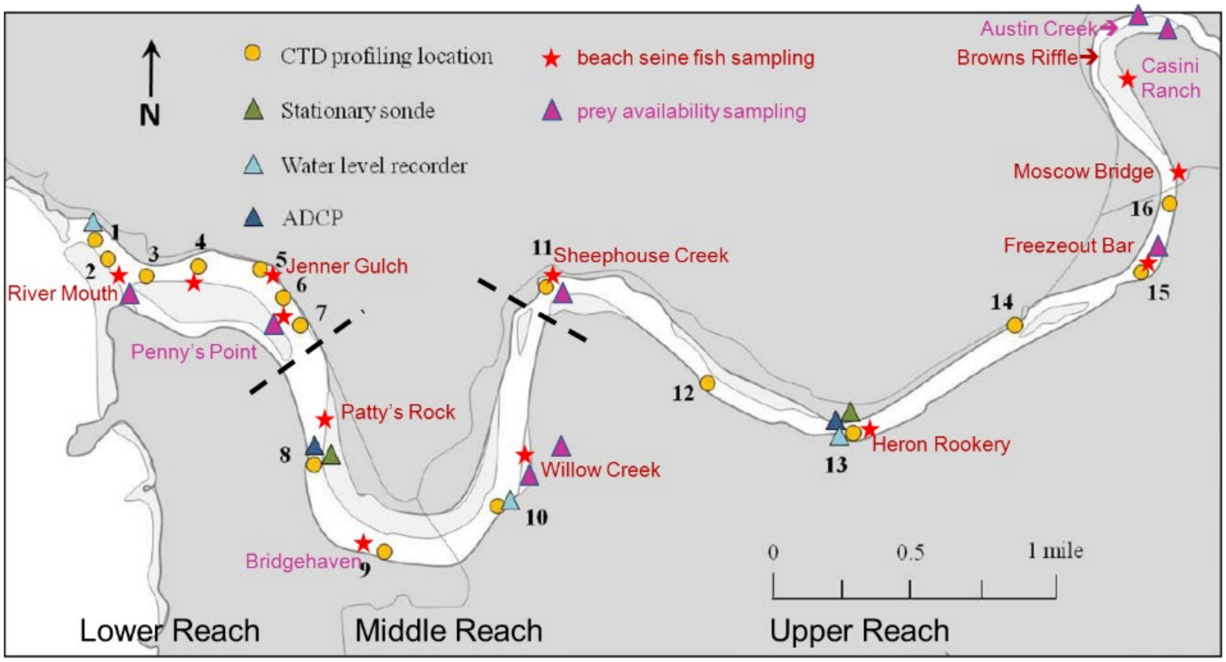


Figure 4.3 1. Locations of sampling stations for juvenile salmon diet (seining location) and prey resource availability (benthic infauna, epibenthos, zooplankton) in three reaches of the Russian River Estuary. Prey availability sampling was conducted at River Mouth, Penny Point, Willow Creek Bar, and Freezeout Bar.

Prey resource availability sampling occurs at four sites distributed through the three estuarine reaches (Figure 4.3.1): Lower Reach—River Mouth and Penny Point; Middle Reach—Willow Creek; and Upper Reach—Freezeout Bar. Each of the sites includes three, lateral transects across the Estuary over which four sampling methods were deployed to sample availability of juvenile steelhead prey (Figures 4.3.2-4.3.6; see Figure 4.3.1 for more specific locations by different sampling methods).

Juvenile Salmon Diet Composition

Systematic sampling of the diets of five or more (n≥5) juvenile steelhead ≥55 mm FL are derived, when available, from the Water Agency beach seine sampling during the Lagoon Management Period between May 15 and October 15. All fish designated for diet analysis are handled, gastric lavaged and released according to the University of Washington animal care protocols. If resources are available and sample sizes are less than five individual fish (n=<5) during systematic sampling, event sampling around scheduled beach management at the barrier beach are coordinated with Water Agency fisheries monitoring and physical measurements of estuarine response.

Stomach lavage follows Foster (1977) and Light *et al* (1983). Diet contents are preserved in 10% Formalin for later laboratory processing. As per Water Agency fisheries protocols, fork lengths and weights are taken from each fish. Each fish is scanned for a passive integrated transponder (PIT) tag and tagged if no previous PIT tag was detected.

Figure 4.3.29. Distribution of juvenile salmonid prey resource availability in three reaches of the Russian River Estuary.

Figure 4.3.3. Distribution of juvenile salmonid prey availability sampling transects and techniques at the River Mouth site in the Russian River Estuary. Fallout trapping for terrestrial invertebrates was conducted prior to 2015.

Figure 4.3 4. Distribution of juvenile salmonid prey availability sampling transects and techniques at the Penny Point site in the Russian River Estuary. Fallout trapping for terrestrial invertebrates was conducted prior to 2015.

Figure 4.3.5. Distribution of juvenile salmonid prey availability sampling transects and techniques at the Willow Creek site in the Russian River Estuary. Fallout trapping for terrestrial invertebrates was conducted prior to 2015.

Figure 4.3.6. Distribution of juvenile salmonid prey availability sampling transects and techniques at the Freezeout Bar site in the Russian River Estuary. Fallout trapping for terrestrial invertebrates was conducted prior to 2015.

In the analysis of 2014 and 2015 fish diet collections, priority of sample processing was based on juvenile steelhead samples that overlapped with the hydroacoustic telemetry monitoring of tagged steelhead. Focusing on diet composition and consumption rate of these selected fish provided the maximum overlap for bioenergetic model estimation of potential growth using the combination of the empirical diet data for fish at the same time and in the same reaches as the thermal regime of the tagged fish.

Prey Resource Availability

Benthic infauna and epibenthos prey resource sampling were conducted once per month in the Lagoon Management Period during open, tidal (baseline) conditions. If barrier beach conditions result in a closure, epibenthos and benthic infauna are sampled seven and 14 days after closure. Following an extended closure of 14 days or more, prey resource availability sampling of benthic infauna, epibenthos, and zooplankton will begin at day 14 and continue every three weeks after until the Estuary opens. Under Estuary conditions in 2014, 696 individual samples were collected (Table 4.3.1); in 2015, 976 individual samples were acquired (Table 4.3.2).

Table 4.3.1. Prey resource availability samples collected in 2014, Russian River Estuary.

Date	Mouth Condition	Jenner Gage Water Level (ft) (10am-2pm)	Benthic Core	Sled Channel	Epi-Benthic Net to Shore	Zooplankton Net
River Mouth						
6/3/2014	OPEN	Gauge Down	12	9	5	3
7/1/2014	OPEN	1.1-1.5	12	9	5	3
7/29/2014	OPEN	0.2-2.1	12	9	5	3
8/26/2014	OPEN	0.6-2.4	12	9	5	3
9/23/2014	CLOSED (6th day of closure	4.2	12	9	5	3
10/9/2014	CLOSED (22nd day of closure)	6.7	12	9	5	3
Penny Point						
6/3/2014	OPEN	Gauge Down	12	9	5	3
7/1/2014	OPEN	1.1-1.5	12	9	5	3
7/29/2014	OPEN	0.2-2.1	12	9	5	3
8/26/2014	OPEN	0.6-2.4	12	9	5	3
9/23/2014	CLOSED (6th day of closure	4.2	12	9	5	3
10/9/2014	CLOSED (22nd day of closure)	6.7	12	9	5	3
Willow Creek						
6/3/2014	OPEN	Gauge Down	12	9	5	3
7/1/2014	OPEN	1.1-1.5	12	9	5	3
7/29/2014	OPEN	0.2-2.1	12	9	5	3
8/26/2014	OPEN	0.6-2.4	12	9	5	3
9/23/2014	CLOSED (6th day of closure	4.2	12	9	5	3
10/9/2014	CLOSED (22nd day of closure)	6.7	12	9	5	3
Freezeout Bar						
6/3/2014	OPEN	Gauge Down	12	9	5	3
7/1/2014	OPEN	1.1-1.5	12	9	5	3
7/29/2014	OPEN	0.2-2.1	12	9	5	3
8/26/2014	OPEN	0.6-2.4	12	9	5	3
9/23/2014	CLOSED (6th day of closure	4.2	12	9	5	3
10/9/2014	CLOSED (22nd day of closure)	6.7	12	9	5	3
Subtotal by sample type			**288**	**216**	**120**	**72**
Total Number of Samples						**696**

Table 4.3.2. Prey resource availability samples collected in 2015, Russian River Estuary.

Date	Mouth Condition	Jenner Gage Water Level (ft) (10am-2pm)	Benthic Core	Sled Channel	Epi-Benthic Net to Shore	Zooplankton Net
River Mouth						
5/27/2015	OPEN	0.8-1.4	12	9	5	3
6/10/2015	CLOSED (12th day of closure)	6.6	12	12	5	3
6/30/2015	OPEN	0.9-1.6	12	9	5	3
7/29/2015	OPEN	1.43-2.1	12	9	5	3
8/26/2015	OPEN	1.0-1.8	12	9	5	3
9/14/2015	CLOSED (7th day of closure)	3.9	12	12	5	3
9/21/2015	CLOSED (14th day of closure)	5.5	12	12	5	3
10/19/2015	CLOSED (9th day of closure)	5.4	12	12	5	3
Penny Point						
5/27/2015	OPEN	0.8-1.4	12	9	5	3
6/10/2015	CLOSED (12th day of closure)	6.6	12	12	5	3
6/30/2015	OPEN	0.9-1.6	12	9	5	3
7/29/2015	OPEN	1.43-2.1	12	9	5	3
8/26/2015	OPEN	1.0-1.8	12	9	5	3
9/14/2015	CLOSED (7th day of closure)	3.9	12	12	5	3
9/21/2015	CLOSED (14th day of closure)	5.5	12	12	5	3
10/19/2015	CLOSED (9th day of closure)	5.4	12	12	5	3
Willow Creek						
5/27/2015	OPEN	0.8-1.4	12	9	5	3
6/10/2015	CLOSED (12th day of closure)	6.6	12	12	5	3
6/30/2015	OPEN	0.9-1.6	12	9	5	3
7/29/2015	OPEN	1.43-2.1	12	9	5	3
8/26/2015	OPEN	1.0-1.8	12	9	5	3
9/14/2015	CLOSED (7th day of closure)	3.9	12	12	5	3
9/21/2015	CLOSED (14th day of closure)	5.5	12	12	5	3
10/19/2015	CLOSED (9th day of closure)	5.4	12	12	5	3
Freezeout Bar						
5/27/2015	OPEN	0.8-1.4	12	9	5	3
6/10/2015	CLOSED (12th day of closure)	6.6	12	12	5	3
6/30/2015	OPEN	0.9-1.6	12	9	5	3
7/29/2015	OPEN	1.43-2.1	12	9	5	3
8/26/2015	OPEN	1.0-1.8	12	9	5	3
9/14/2015	CLOSED (7th day of closure)	3.9	12	12	5	3
9/21/2015	CLOSED (14th day of closure)	5.5	12	12	5	3
10/19/2015	CLOSED (9th day of closure)	5.4	12	12	5	3

Date	Mouth Condition	Jenner Gage Water Level (ft) (10am-2pm)	Benthic Core	Sled Channel	Epi-Benthic Net to Shore	Zooplankton Net
Subtotal by sample type			**384**	**336**	**160**	**96**
Total Number of Samples						**976**

Benthic Infauna

Replicate core samples (0.0024-m^2 PVC core inserted 10 cm in to the sediment) are taken at each transect of each site. The location of each core sample is consistent with each epibenthic sled and epibenthic net to shore sample, but no core samples are taken in between transects. This sample is repeated four times per transect (twelve times per site). Additional samples would be added along the transect with increasing water level (inundation of the shoreline) during closure or outlet channel implementation. The sediment cores are preserved in 10% buffered Formalin for laboratory analysis.

Epibenthos

Epibenthic organisms at the sediment-water interface are sampled with two methods: (1) epibenthic net (net to shore); and, (2) epibenthic (channel) sled. The epibenthic net is a 0.5-m x 0.25-m rectangular net, equipped with 106-µm Nitex mesh that is designed to ride along the surface of the Estuary bottom substrate. It is deployed 10 m from shore and then pulled along the bottom perpendicular back to shore by an individual onshore. This is replicated five times per site (once at each transect and then once between Transects 1 and 2 and also between Transects 2 and 3). The epibenthic sled is equipped with a 0.125-m^2 opening, 1-m long 500-µm Nitex mesh net towed behind the boat against the current. The sled is dropped off of the bow of the boat and allowed to sink to the bottom. Once the boat has finished towing the sled (in reverse) 10 m against the current, it will be retrieved back onto the boat. This is replicated five times per site (once at each transect and then once between Transects 1 and 2 and also between Transects 2 and 3). The sled is used to obtain three samples per transect (nine per site under open conditions). Additional samples would be added along the shoreward margin of the transect with increasing water level (inundation of the shoreline) during closure or outlet channel implementation. Captured organisms are preserved in 10% buffered Formalin for laboratory analysis.

Zooplankton

Zooplankton are sampled at the same location as water quality (the deepest available depth per site) using a 0.33-m diameter ring net, 73-µm Nitex mesh and cod end cup. Replicated (n=3) vertical water column hauls are made by lowering the zooplankton net until the top ring of the net is just above the benthos and then pulled by hand vertically to the surface to obtain a sample of the entire water column. This sample set is repeated three times per site. Captured organisms are preserved in 10% buffered Formalin for laboratory analysis.

Sample Processing and Analyses

Stomach contents from juvenile salmon are identified to the species level if possible under a dissecting microscope. Invertebrates found in the diets of steelhead and collected in the prey resource samples are identified to species level, except for insects which are identified to family level. Any invertebrate collected during prey sampling and not found to be part of the steelhead diet is identified to order or family level. Each of the identified prey taxa are counted (for numerical composition) and weighed (for gravimetric [biomass] composition) and the frequency of occurrence. The state of total stomach content biomass is normalized by individual fish weight to provide an additional index of relative consumption rate ("instantaneous" ration), which is the total biomass of prey found in individual fish stomach contents relative to the biomass of the fish expressed as g g^{-1}. It is recognized that this is only a short-term index of consumption, and will vary by fish size, time of day and other factors influencing foraging behavior. If fish are captured under the same general conditions, this index can provide an indication of differences in feeding performance. Under some conditions, the instantaneous ration can be used to develop an estimate of daily ration that can be used in bioenergetic modeling of potential growth.

In addition to individual metrics of diet composition, the Index of Relative Importance (IRI; Pinkas et al. 1971) is also calculated, wherein %Total IRI for each discrete prey taxa takes into account the proportion that prey taxa constitutes of the total number and biomass of prey and the frequency of occurrence of that taxa among in the total number of fish stomach samples:

$$IRI_i = FO_i*[NC_i + GC_i]$$

where NC is the percent numerical composition, GC is the percent gravimetric (biomass) contribution, FO is the percent frequency of occurrence for each of the prey taxa, and *i* is the prey taxa; results are expressed as a percentage of the total IRI for all prey items. We also interpret diet composition using just GC_i in order to better represent the bioenergetic contribution of prominent (from a FO_i standpoint) prey.

In accordance with a more recent revision of the IRI index, we calculated the Prey-Specific Index of Relative Importance (PSIRI) which substitutes NC and GC with their corresponding prey-specific abundances, %PNC and %PGC:

$$PSIRI_i = FO_i*[\%PNC_i + \%PGC_i]$$

PSIRI sums to 200% and therefore diving by 2 results in a version of the standardized %IRI (Amundsen et al. 1996; Cortẽs 1997), with an important distinction: the PSIRI is additive with respect to taxonomic levels, such that the sum of PSIRI for species will be equal to the PSIRI of the family containing those species.

Prey availability data are standardized to density per area or volume, i.e., m^2 for benthos and epibenthos and m^3 for zooplankton. Prior to analysis, density data are square root transformed to better equate group variances and compress positively skewed distributions to a more nearly normal distribution. All taxa recorded from all samples appears in the raw and processed databases; however, to focus our assessment of Estuary condition effects on prey availability, we confined our graphical and multivariate analyses to identified prey of juvenile steelhead. Furthermore, benthic and epibenthic fauna, taxa that contributed at least 5% of the density to anyone sample, and copepod nauplii, tintinnids, and rotifers were removed from the data used for analysis. Copepod nauplii, tintinnids and rotifers were removed mainly for clarity in viewing the graphs, but they are also because these small plankters tend to be artificially abundant when the zooplankton net becomes clogged with filamentous algae. They are also all too small to show up as prey in planktivorous fish, except for the smallest of larval fish.

Multivariate analyses are also utilized to organize fish diet sample compositions and prey availability samples into statistically distinct groupings. All statistical analyses are performed using the PRIMER v6.0 multivariate statistics analysis package (Clarke and Gorley 2006). The primary analyses included non-metric multidimensional scaling (NMDS) and associated analyses of similarities (ANOSIM) and similarity percentages (SIMPER) of factors (in this case, organism taxa) that account for the similarity. Similarity is based on the Bray-Curtis similarity coefficient. The primary ANOSIM statistic for differences between groups is the Global R, which varies between 0 (no significant difference) to 1 (maximum difference). These analytical tools, and the PRIMER package in particular, are used extensively in applied ecology and other scientific inquiries where the degree of similarity in organization of multivariate data (e.g., species, ecosystem attributes) is of interest.

Results

Estuary Conditions

The Estuary did not experience mouth closure in 2014 until mid-September (Figure 4.3.7). As a result, most of the fish diet and prey availability samples did not occur in or bracket the late closure period. The two scheduled prey availability sampling events on 23 September and 9-10 October did provide some indication of potential effects of Estuary closure but not with comparable samples during recent open conditions. In contrast, the Estuary underwent three closures during 2015 (Figure 4.3.8), the first of which was 16 days in duration between 29 May and 14 June and which was bracketed by prey availability sampling. Later, longer closures occurred from mid-September to mid-October and initiated again soon thereafter.

Juvenile Steelhead Diet Composition

Between 2009 and 2015 a total of 509 juvenile steelhead diets have been sampled for diet composition and consumption rate (Table 4.3.1). Only 74 were sampled in 2014 and 2015, with 29 occurring during a closed inlet state and 45 occurring during an open inlet state (Figures 4.3.7 and 4.3.8). The composition of juvenile steelhead diets through 2014 and 2015 was fairly consistent with previous years of sampling, wherein epibenthic crustaceans—the gammarid amphipods *Eogammarus confervicolus* and *Americorophium* spp., and the isopod *Gnorimosphaeroma insulare*—dominated the numerical and gravimetric composition and occurred in greater than 45% of the samples (Figure 4.3.9). Similar to previous years, corixid beetles (water boatman),

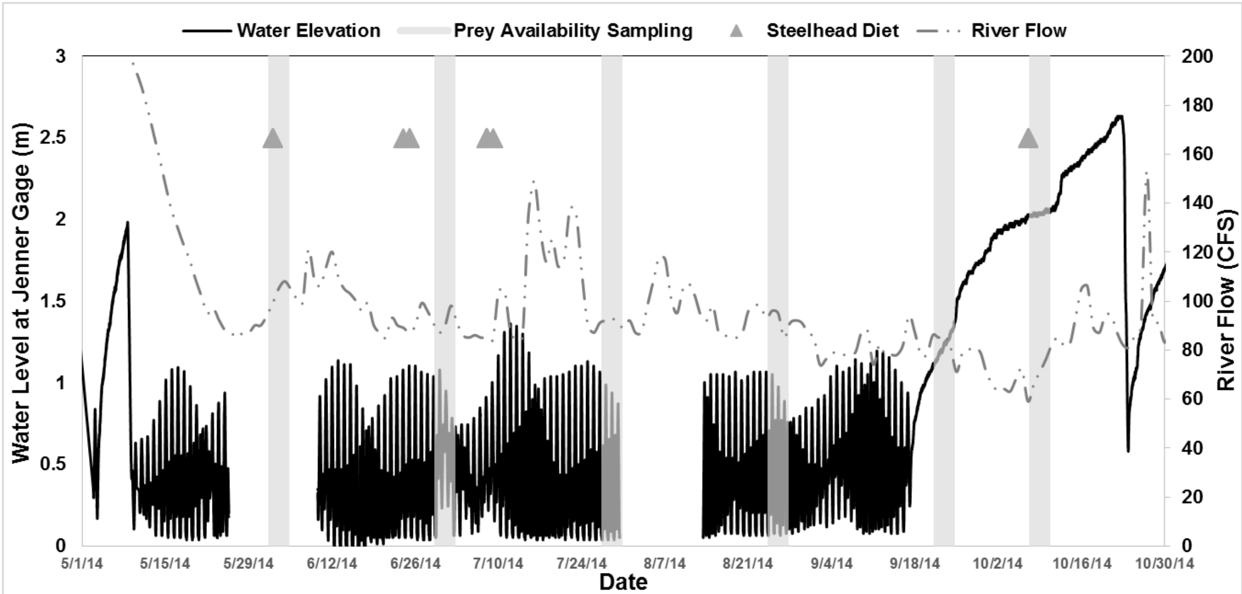


Figure 4.3.7. Water level height (m) at the Jenner Gage from May 1-October 30, 2014, with coincident river inflow (USGS, CFS), steelhead diets, and invertebrate sampling.

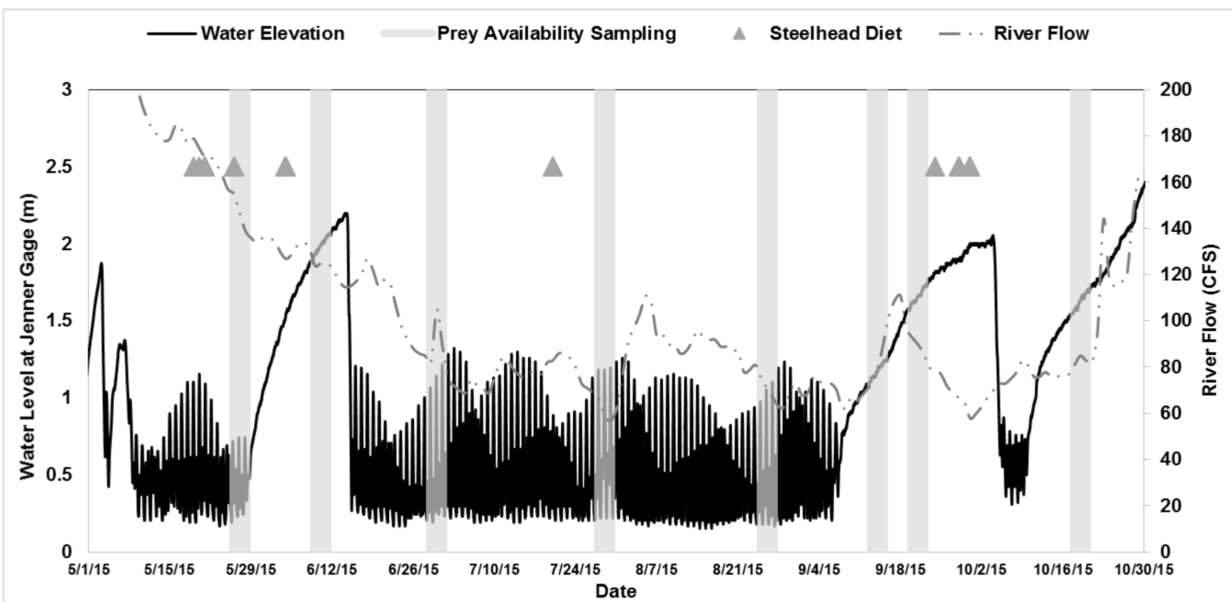


Figure 4.3.8. Water level height (m) at the Jenner Gage from May 1-October 30, 2015, with coincident river inflow (USGS, CFS), steelhead diets, and invertebrate sampling.

Table 4.3.3. Sources of juvenile steelhead samples for diet composition and consumption rate; size range (FL mm) in parentheses.

	Closed			Open		
Year	Lower	Middle	Upper	Lower	Middle	Upper
2009		3 (124-208)		8 (89-113)	41 (56-296)	53 (56-250)
2010	9 (153-210)		4 (160-293)	45 (63-238)	57 (59-235)	127 (65-235)
2011	1 (126)		7 (136-227)	35 (103-325)	19 (94-288)	3 (157-172)
2013		1 (85)	5 (60-168)			16 (61-185)
2014		4 (117-261)	2 (172-257)	11 (73-193)	17 (64-179)	1 (93)
2015	2 (258-303)	4 (184-289)	17 (64-241)		1 (162)	15 (71-191)

juvenile chironomids (midges), the estuarine mysid *Neomysis mercedis* and various insects also appeared as supplementary prey.

Several divergences in diet composition from previous years included higher representations by adult midges and gastropods (snails), and the first occurrence of shore crabs (Decapoda), in 2015. A large portion of the numerical composition during the June 2015 closure consisted of adult midges (Figure 4.3.9). It is important to note that this closure was the earliest that juvenile steelhead diets had been sampled since 2009 and these fish were captured the furthest upstream out of all the capture sites (Brown's Riffle). In addition to the earlier closure, we also observed differences in diet composition during late season sampling (Figure 4.3.10). One of these anomalies was the occurrence of six shore crabs consumed by two larger steelhead (212-303 mm FL) during the September/October 2015 closure; although rare in occurrence, their large mass was notable. During the same closure, we found that approximately 50% (322 out of 650) of the gastropods (suspected non-indigenous New Zealand mudsnail, *Potmopyrgus antipodarum*) in the overall diet composition were consumed by only four of the 509 individuals sampled for diets. One of these fish (241 mm FL) consumed 295 of these snails alone.

Overall, there tends to be a higher diversity of prey consumed earlier in the sampling period than later. This can be observed in the 2014, 2015, and the entire study (2009-2015) numerical composition (Figures 4.3.10-4.3.12) where fewer prey taxa compose a higher proportion of the overall diet later in the sampling period.

Although the difference was small, there was a significant difference detected between all reaches overall (Figure 4.3.13; Table 4.3.4), much of which could be attributed to the occurrence of insects and corixid beetles in the upper reach and the greater occurrence of mysids in the lower reach. There was also a difference detected between open and closed conditions (Figure 4.3.14; Table 4.3.5), but this difference was minor (ANOSIM: $R=0.104$, $P>0.028$). Interestingly, the only difference detected between two reaches during closed conditions was between the middle and lower reaches (ANOSIM: $R=0.197$, $P<0.05$) but all three

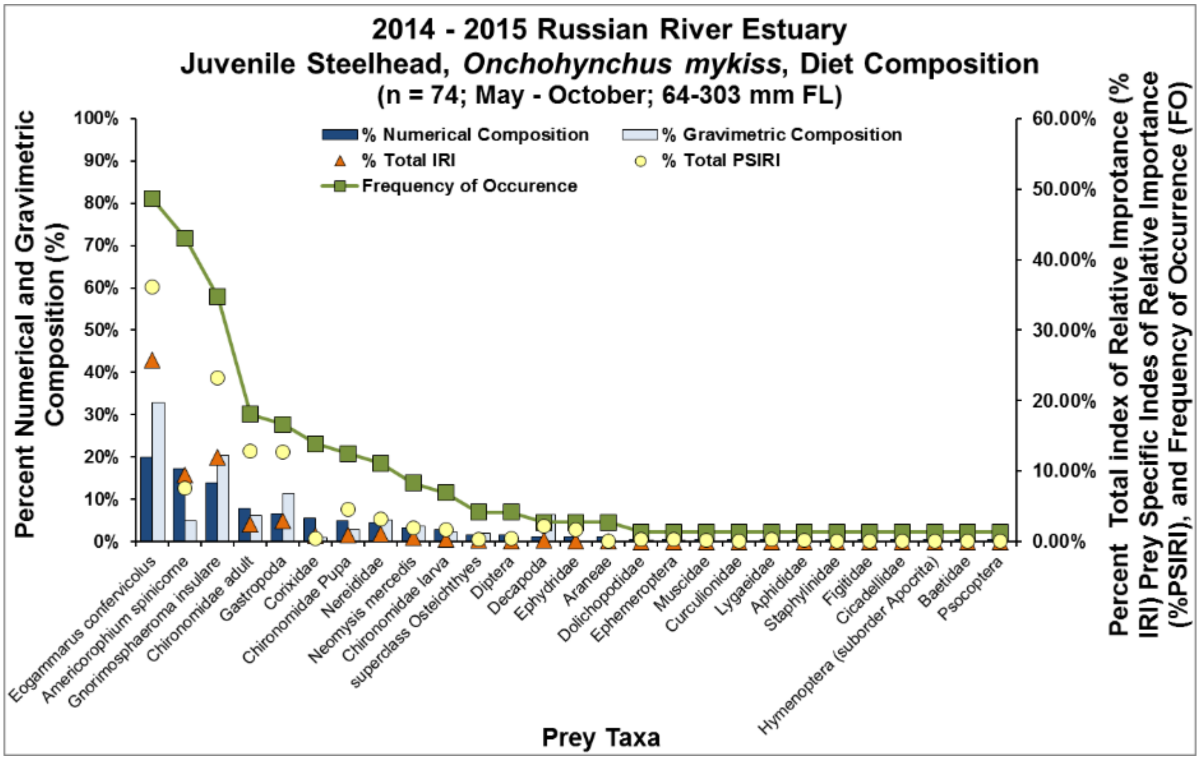


Figure 4.3.9. Percent numerical (NC) and gravimetric composition (GC), frequency of occurrence (FO), total Percent Index of Relative Importance (%IRI) and Prey-Specific Index of Relative Importance (%PSIRI) of prey taxa consumed by juvenile steelhead in the Russian River Estuary, May-October 2014-2015.

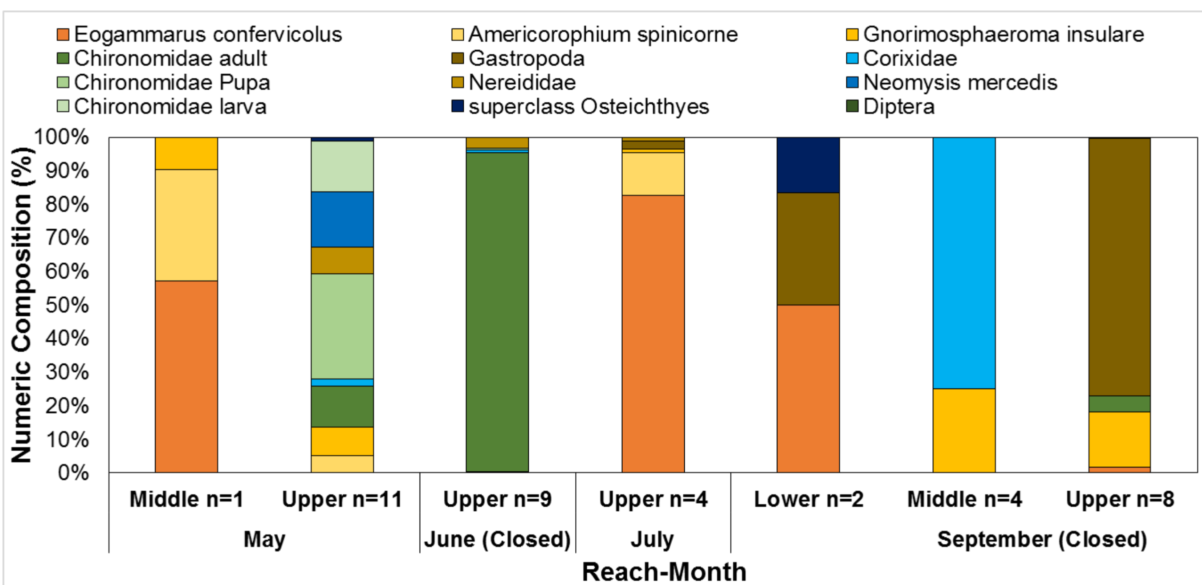


Figure 4.3.10. Percent numerical diet composition of juvenile steelhead in lower, middle and upper reaches of the Russian River Estuary, May-September 2015.

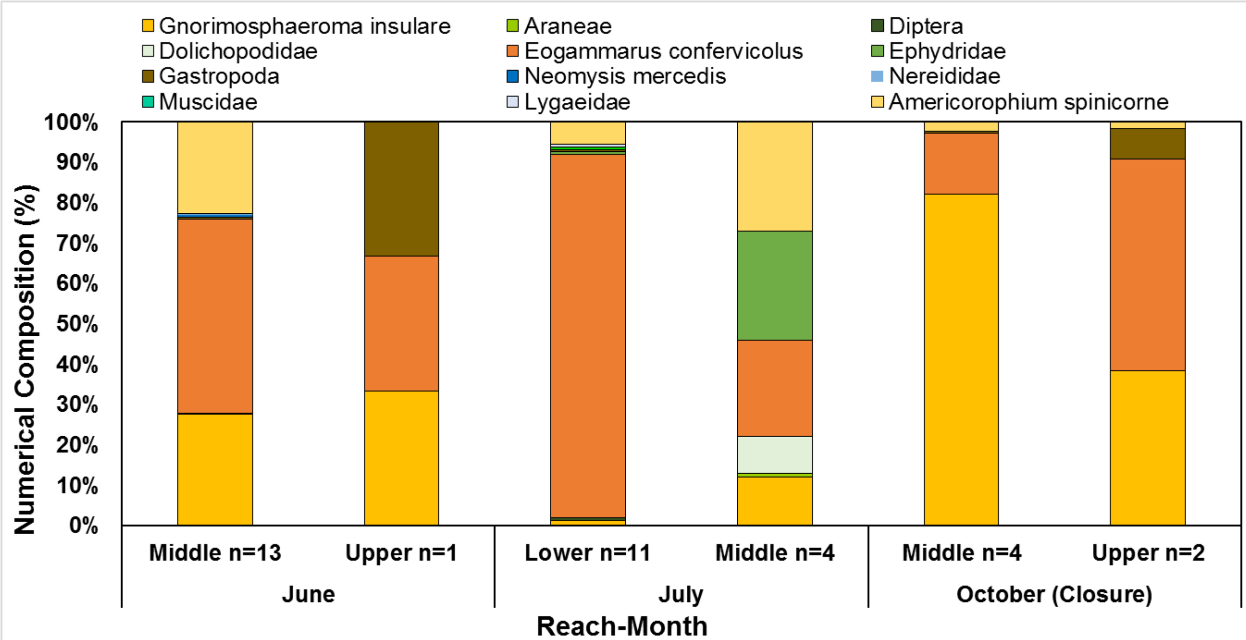


Figure 4.3.11. Percent numeric diet composition of juvenile steelhead in lower, middle and upper reaches of the Russian River Estuary, June-October 2014.

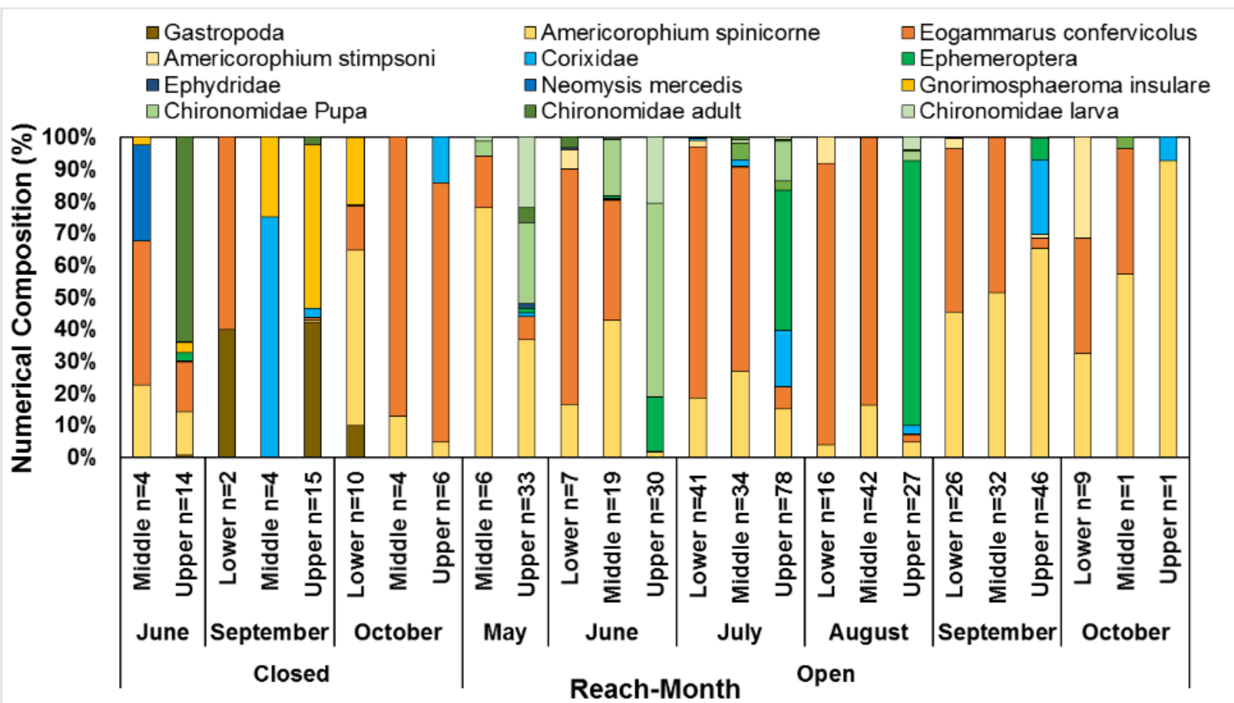


Figure 4.3.12. Percent numeric diet composition of juvenile steelhead in lower, middle and upper reaches of the Russian River Estuary, May-October 2009 - 2015.

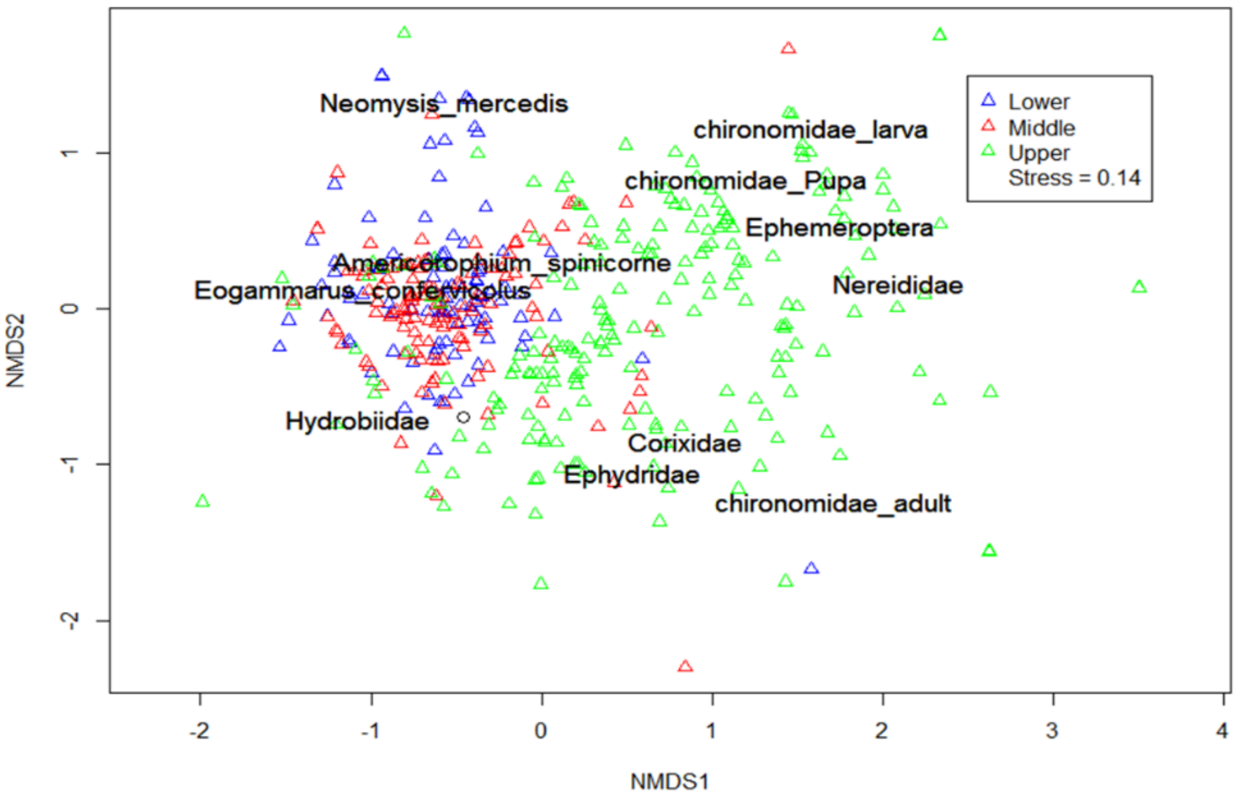


Figure 4.3.13. Multivariate analysis (NMDS) diagram of juvenile steelhead diet composition in lower, middle and upper reaches of the Russian River Estuary, 2009-2015.

Table 4.3.4. Results (R statistic) of ANOSIM multivariate analysis of differences in juvenile steelhead diet composition among lower, middle and upper reaches of the Russian River Estuary, 2009-2015; P values in parentheses where bold values indicate significant differences.

	Lower	Middle	Upper
Lower		0.019 **(0.043)**	0.1356 **(0.001)**
Middle			0.143 **(0.001)**
Upper			

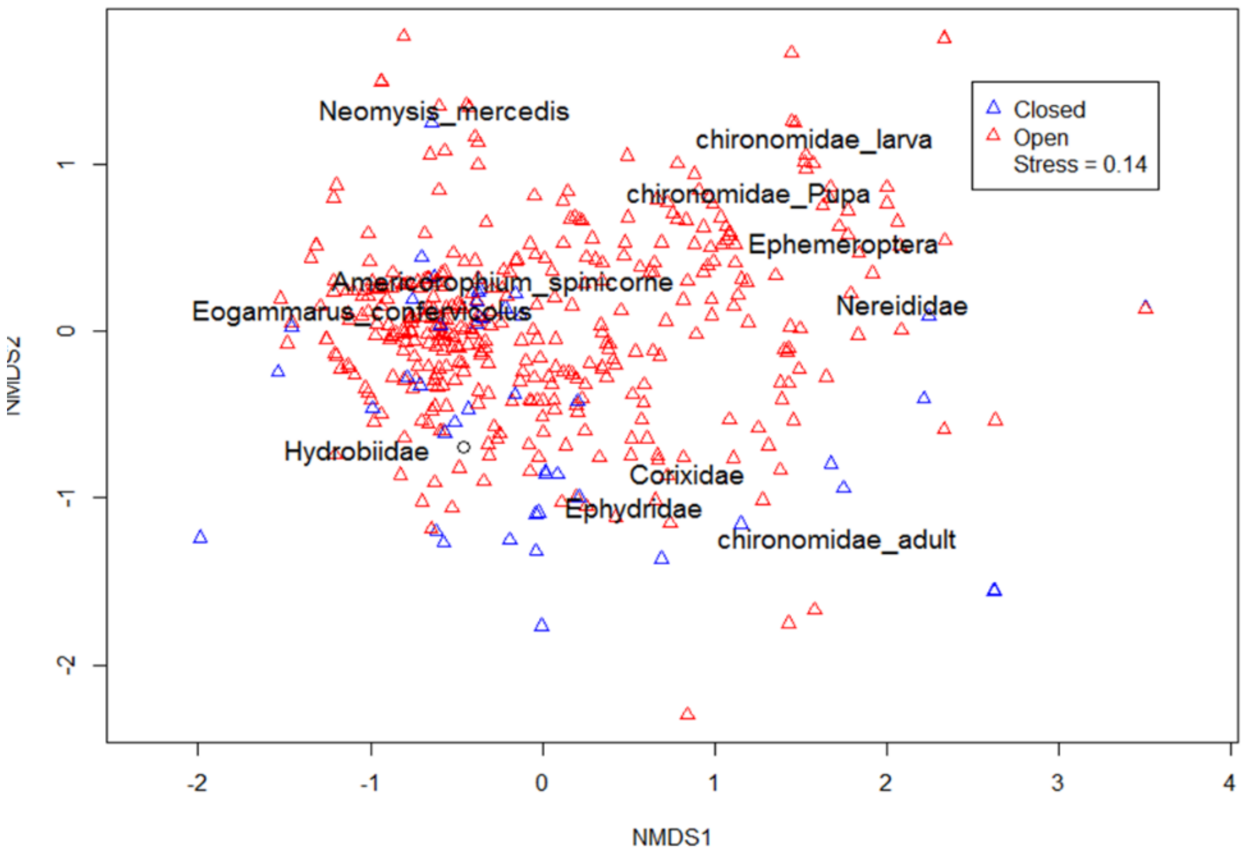


Figure 4.3.14. Multivariate analysis (NMDS) diagram of juvenile steelhead diet composition under open and closed mouth conditions in lower, middle and upper reaches of the Russian River Estuary, 2009-2015.

Table 4.3.5. Results (R statistic) of ANOSIM multivariate analysis of differences in juvenile steelhead diet composition among lower, middle and upper reaches of the Russian River Estuary, 2009-2015; P values in parentheses where bold values indicate significant differences.

		Closed			Open		
		Lower	Middle	Upper	Lower	Middle	Upper
Closed	**Lower**		0.1967 **(0.008)**	0.02393 (0.604)	0.079 (0.171)	0.1876 (0.064)	0.1305 **(0.037)**
	Middle			0.04054 (0.696)	0.1882 **(0.035)**	0.1235 (0.082)	0.08 (0.114)
	Upper				0.4275 **(0.001)**	0.4854 **(0.001)**	0.2035 **(0.001)**
Open	**Lower**					0.03077 **(0.017)**	0.1929 **(0.001)**
	Middle						0.208 **(0.001)**
	Upper						

reaches were significantly different during open conditions (Table 4.3.5). The largest differences detected were between the upper reach during closed conditions and both the lower and middle reaches during open conditions. The upper reach was the only reach where we detected a difference between open and closed conditions.

Variations of instantaneous ration indices for fish of the same relative size caught in different sites suggest some differences in feeding performance (Figure 4.3.15). The most apparent comparison evidenced by sufficient sample sizes was the apparently higher consumption of fish from Bridgehaven (middle reach) in June as compared to Bridgehaven in July and Jenner Gulch (lower reach) in July.

Prey Availability

Samples collected during the 2014 and 2015 Lagoon Management Periods analyzed by University of Washington were prioritized for contrast in Estuary status/water level and overlap with juvenile steelhead diet analyses from those periods. Benthic samples from 2014 provided a comparison from 3 June (open, no gauge data), 23 September (closed, 4.2 ft), and 10 October (closed, 6.7 ft); channel epibenthic sled and epibenthic net samples included the June and September dates and 9 October (closed, 6.7 ft) samples; and, zooplankton samples from 2014 included the same 3 June, 23 September and 9 October contrast.

Benthic Infauna

Among the prevalent prey of juvenile steelhead in 2015, as described above, the motile amphipod *Eogammarus confervicolus*, tubicolous amphipods *Americorophium* spp. and gastropod snails were most abundant on May 27, before the Estuary closed (Figure 4.3.16). Willow Creek was the location with consistently highest densities of macroinvertebrate prey, with means of 15,000 to 20,000 individuals m^{-2}.

The same prey taxa dominated the benthos assemblage on June 12, 12 days into the closure, but at densities approximately half the densities observed for the same taxa in late May (Figure 4.3.17). By 30 June, when tidal conditions had been prevailing for sixteen days since the Estuary reopened, mean densities of the same taxa were generally reduced by 50% of the June 10 densities. Except for the gastropod snails (which appeared in maximum density at Willow Creek), the greatest densities of most taxa occurred at the River Mouth sampling site at this time (Figure 4.3.18). Multivariate analysis of the taxa density composition among the four sites over the three dates bracketing the Estuary closure (Figure 4.3.19; 2D stress =0.2) indicated no significant difference among either sites (Global R = 0.197) or dates (Global R = 0.084). The most dissimilar benthos composition were between Penny Point and Freezeout Bar (average Bray-Curtis dissimilarity = 71.4), which was contributed by differences in densities of *Anisogammarius* spp., *A. spinicorne* and gastropods (56% of total dissimilarity). The least dissimilar were the benthos samples from River Mouth and Willow Creek (58.8%). Overall, if there was a response to the closure by benthic invertebrate prey, it was likely confined to relative universal reduction in the taxa across all four sites, perhaps as a result of changes in water quality parameters through the closure.

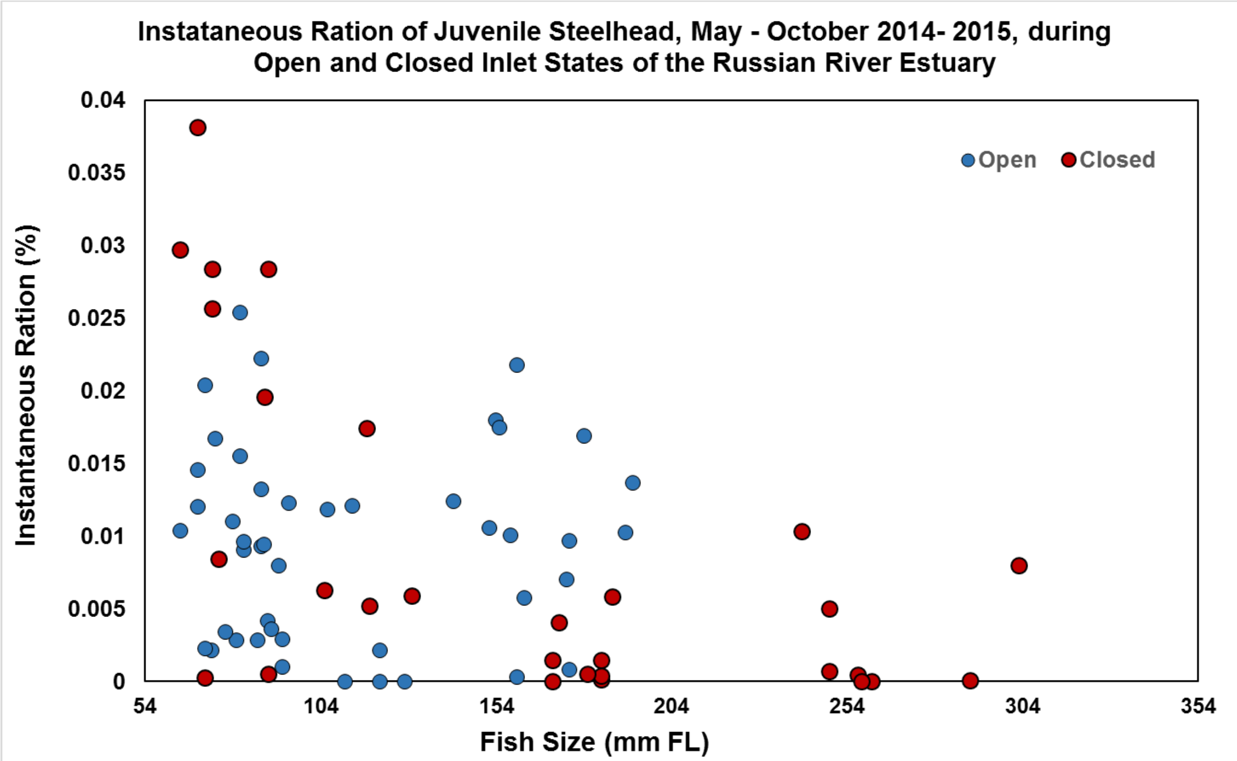


Figure 4.3.15. Instantaneous ration of juvenile steelhead during open and closed conditions in the Russian River Estuary, May-October 2014 - 2015.

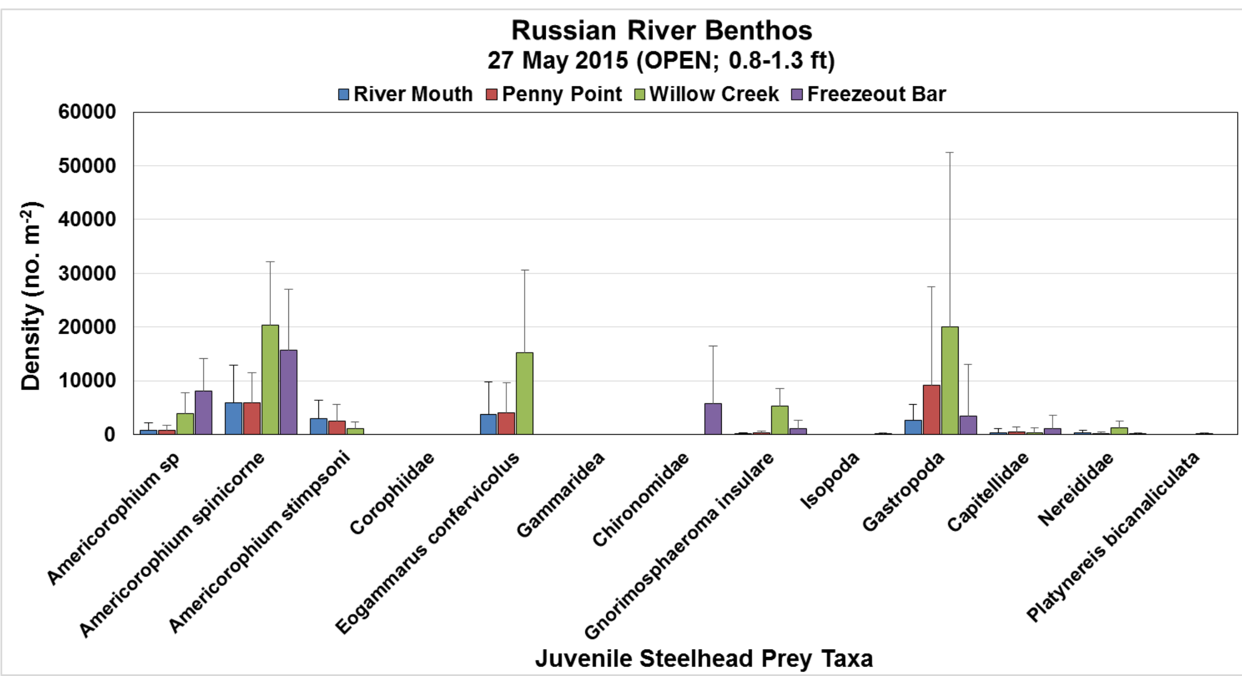


Figure 4.3.16. Density of benthic macroinvertebrates documented as juvenile steelhead prey at four sites in the Russian River Estuary, 27 May 2015.

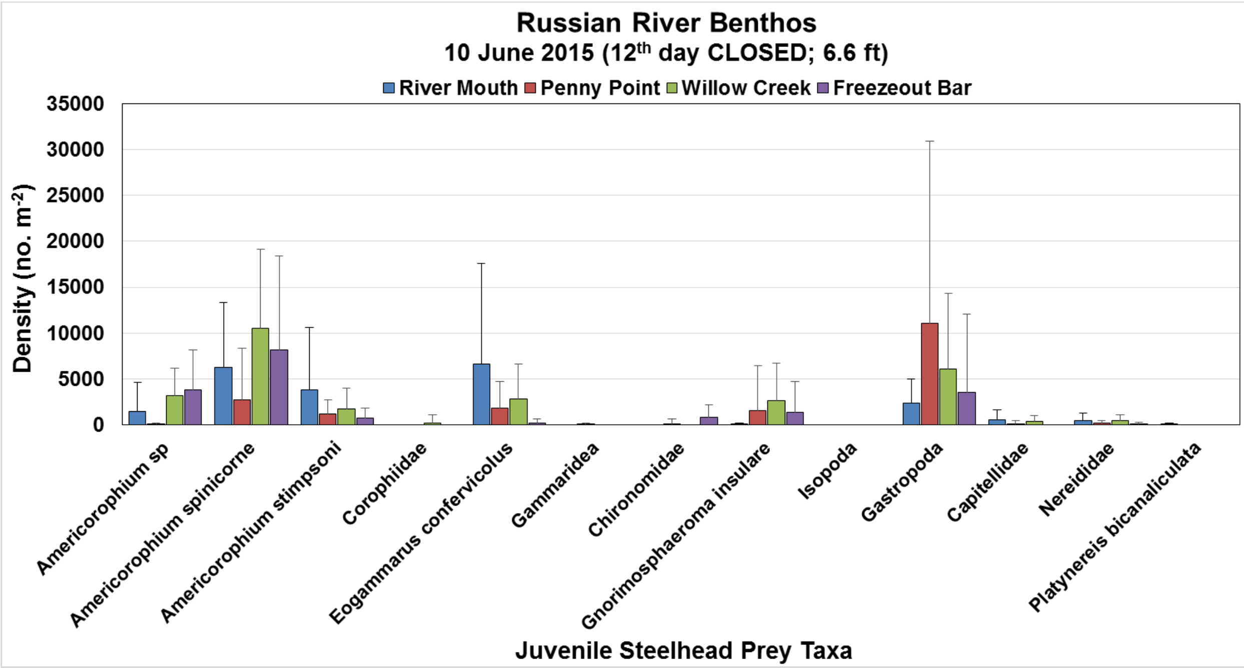


Figure 4.3.17. Density of benthic macroinvertebrates documented as juvenile steelhead prey at four sites in the Russian River Estuary, 10 June 2015.

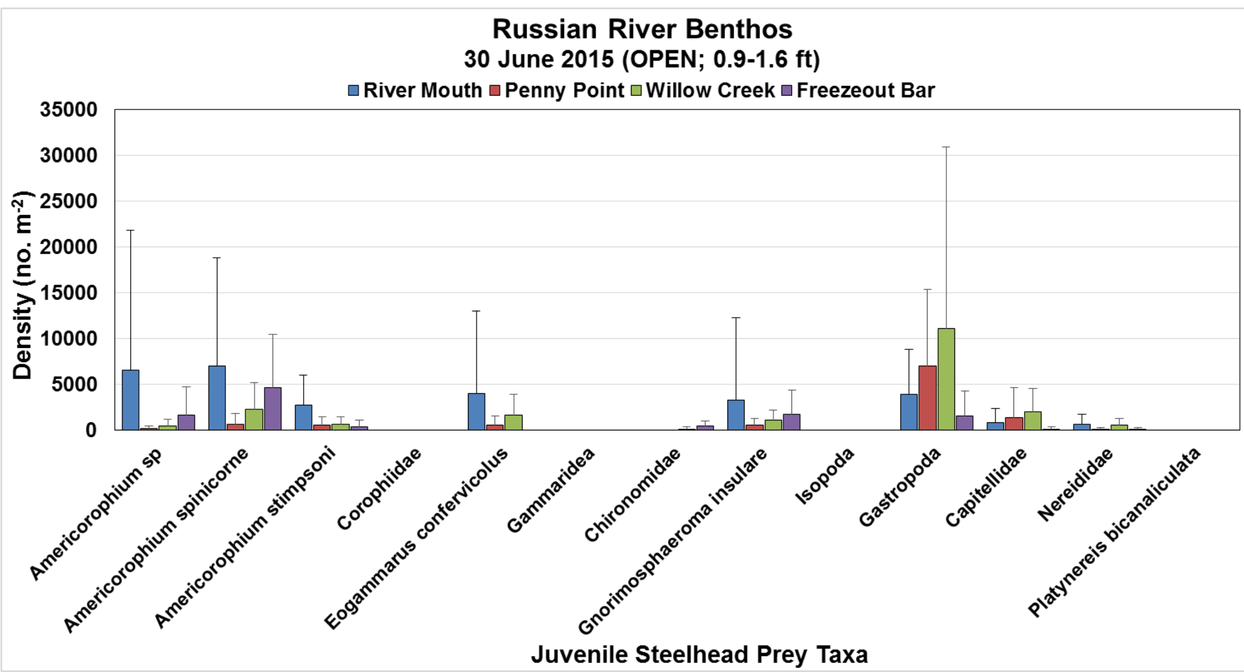


Figure 4.3.18. Density of benthic macroinvertebrates documented as juvenile steelhead prey at four sites in the Russian River Estuary, 30 June 2015.

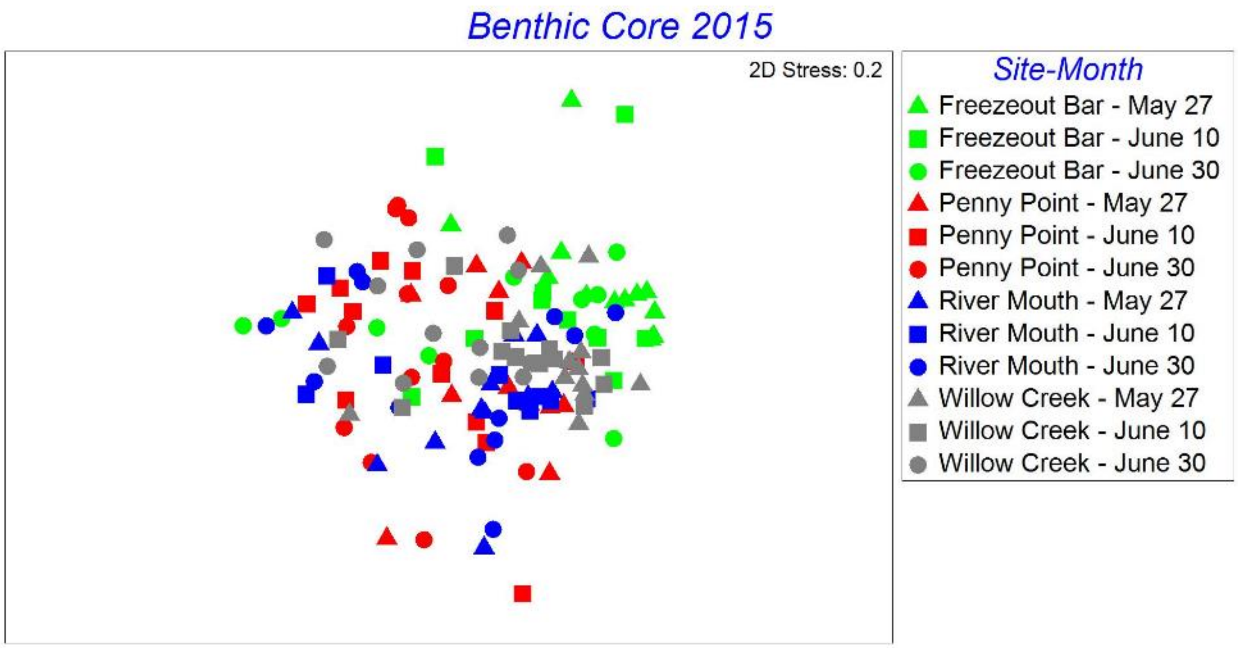


Figure 4.3.19. Multivariate analysis (NMDS) diagram of density composition of benthic macroinvertebrate prey of juvenile steelhead in lower, middle and upper reaches of the Russian River Estuary, 2015.

In comparison, benthic macroinvertebrates in early June 2014 occurred most commonly and abundantly at Penny Point when the Estuary was open (Figure 4.3.20; but note that samples were not available from Willow Creek). *Americorophium spp.* amphipods were similarly dominant, but the epibenthic isopod *Gnorimosphaeroma insulare* was equally dense as well, averaging between ~17,000 and ~27,000 organisms m^{-2}. Similar to 2015, *A. spinicorne* was the most abundant prey taxa occurring in the upper reach, at Freezeout Bay, in comparable abundance (~11,000 m^{-2}).

Somewhat similar patterns in benthic macroinvertebrate prey response to Estuary closure was evident in 2014, although sampling was not bracketed during open periods recent to the closure. The Estuary did not close for any significant period until late September; by the time of prey availability sampling on September 23, it had been closed six days and the water level had risen to 4.2 ft (Table 4.3.1). At this time, the dominant juvenile steelhead prey were more uniformly distributed among all reaches of the Estuary (Figure 4.3.21). *Americorophium* spp. amphipods and *G. insulare* isopods were most abundant, averaging between ~2,000 and ~15,000 organisms m^{-2}, in the lower and middle reaches but less so in the upper reach, at Freezeout Bar. The epibenthic amphipod *Eogammarus confervicolus* had also appeared in average densities of up to ~6,000 m^{-2}, predominantly in the lower and middle reaches. By October 10, 22 days into the closure and the Estuary's water elevation having risen to 6.7 ft, prey availability had diminished at all sites except for Penny Point, were densities of *Americorophium* spp. were still comparable to September 23 but *G. insulare* and *E. confervicolus* had declined (Figure 4.3.22). Multivariate analysis (not presented here) paralleled that found for 2015.

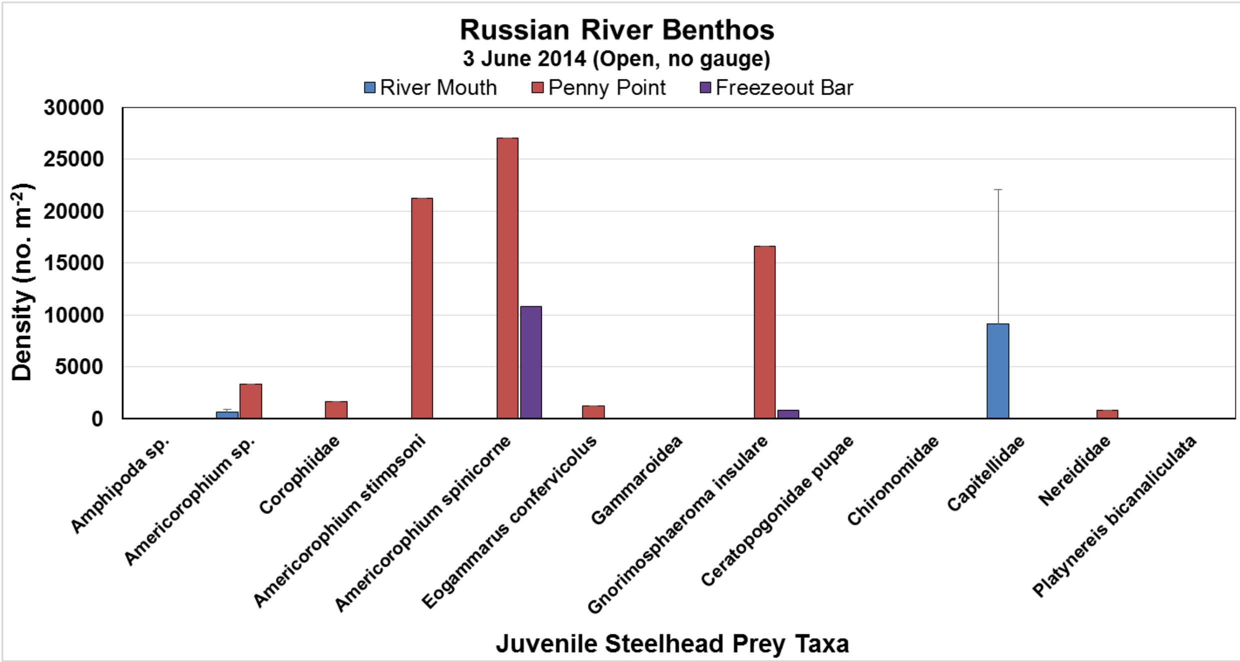


Figure 4.3.20. Density of benthic macroinvertebrates documented as juvenile steelhead prey at three sites in the Russian River Estuary, 3 June 2014.

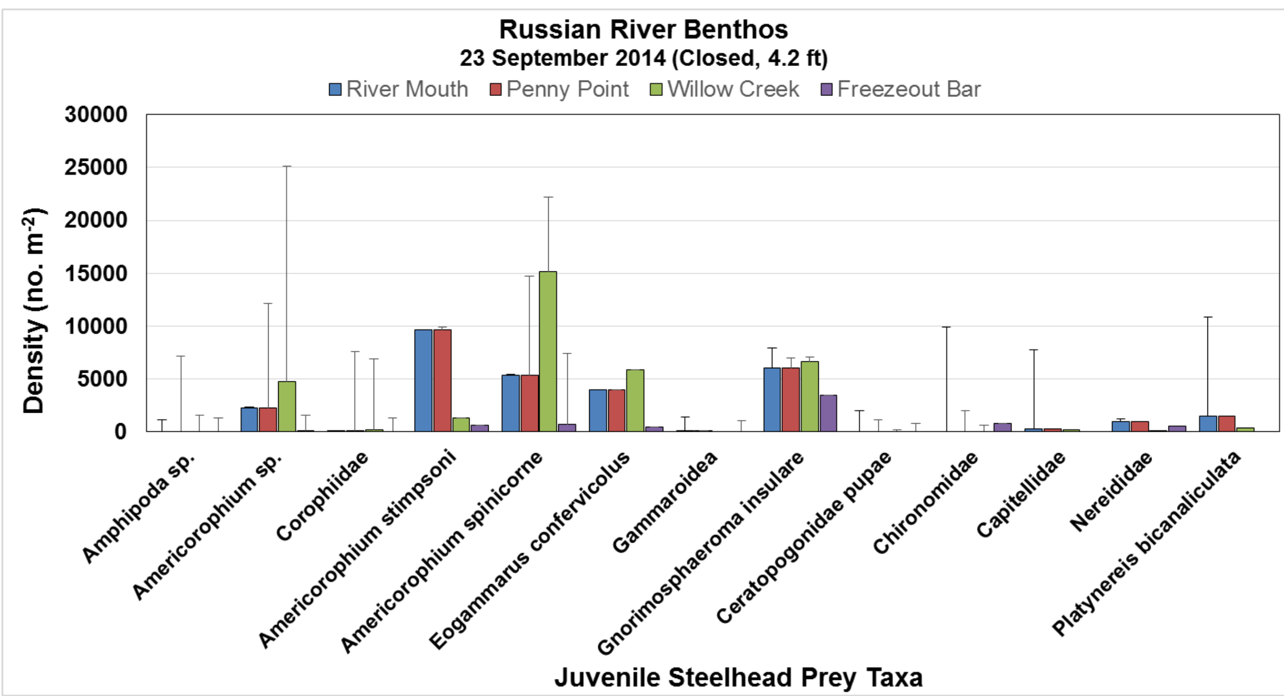


Figure 4.3.21. Density of benthic macroinvertebrates documented as juvenile steelhead prey at four sites in the Russian River Estuary, 23 September 2014.

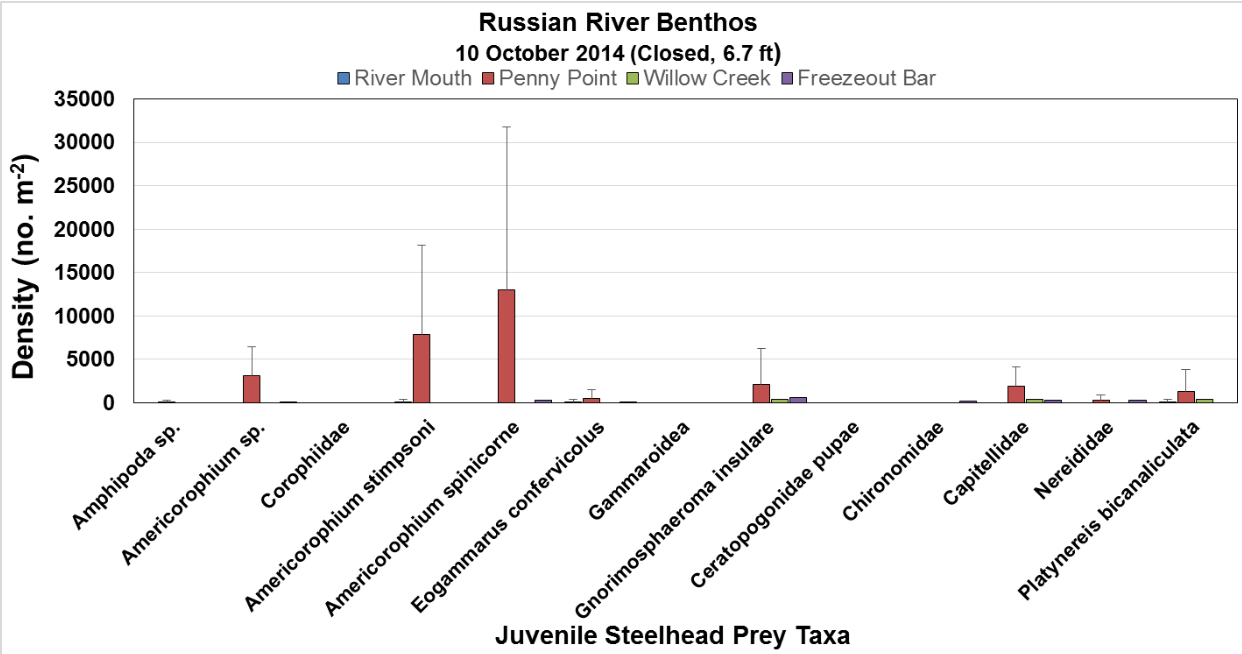


Figure 4.3.22. Density of benthic macroinvertebrates documented as juvenile steelhead prey at four sites in the Russian River Estuary, 10 October 2014.

Epibenthic Net to Shore

As described in Methods, sampling by the epibenthic net samples within 10 m of the high water level could be indicative of a shift in prey organism distribution as a function of Estuary water level and volume. As water elevation rises above 2.1 ft (Jenner Gauge) during a closure event, the epibenthic net sample organisms have migrated into the recently inundated shallow water margin. This may account for the shift from mean densities of ~300-5000 individual m^{-2} *Americorophium* spp. amphipods, *G. insulare* isopods and gastropod snails, primarily at the River Mouth site, in late May 2015 (Figure 4.3.23) to primarily *E. confervicolous* amphipods and gastropod snails mid-way through the closure on 10 June, albeit at lower densities and increased occurrence at Penny Point (Figure 4.3.24). After the Estuary mouth reopened, *Americorophium* spp. amphipods, *G. insulare* and gastropods returned to higher mean densities, ~1400-3100 individual m^{-2}, in late June, again concentrated at the River Mouth site (Figure 4.3.25). Multivariate analysis indicated significant differences (Global R = 0.646) in taxa density compositions among the four sites and somewhat equal (R = 0.490) differences among the dates bracketing the Estuary closure (Figure 4.3.26). As might be expected, ANOSIM analysis indicated that the most significant differences between dates were before-during (R = 0.519) and during-after the closure (R = 0.619) as compared to before-after the closure (R = 0.349). Differences (SIMPER dissimilarity) among density composition was due primarily to varying contributions from the three prominent prey taxa—*Americorophium* spp. and *E. confervicolus* amphipods and the isopod *G. insulare*—but also corixid beetles and their nymphs when comparing Freezeout Bay samples to any other site.

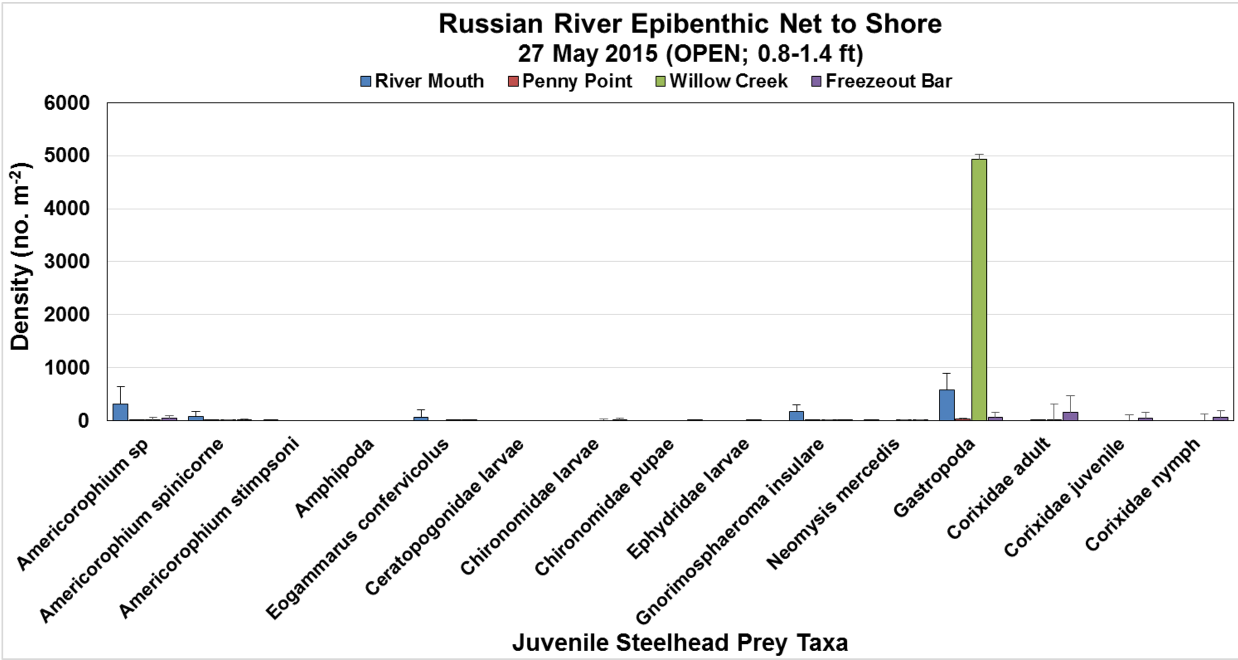


Figure 4.3.23. Density of epibenthic net macroinvertebrates documented as juvenile steelhead prey at four sites in the Russian River Estuary, 27 May 2015.

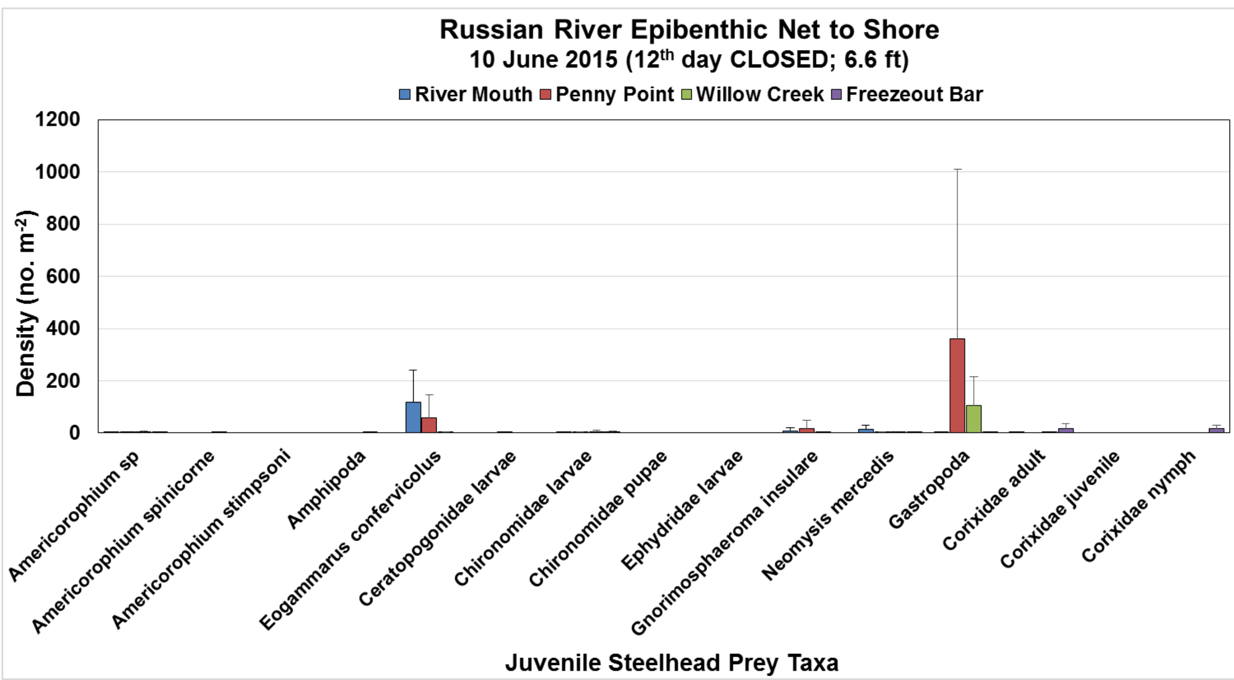


Figure 4.3.24. Density of epibenthic net macroinvertebrates documented as juvenile steelhead prey at four sites in the Russian River Estuary, 10 June 2015.

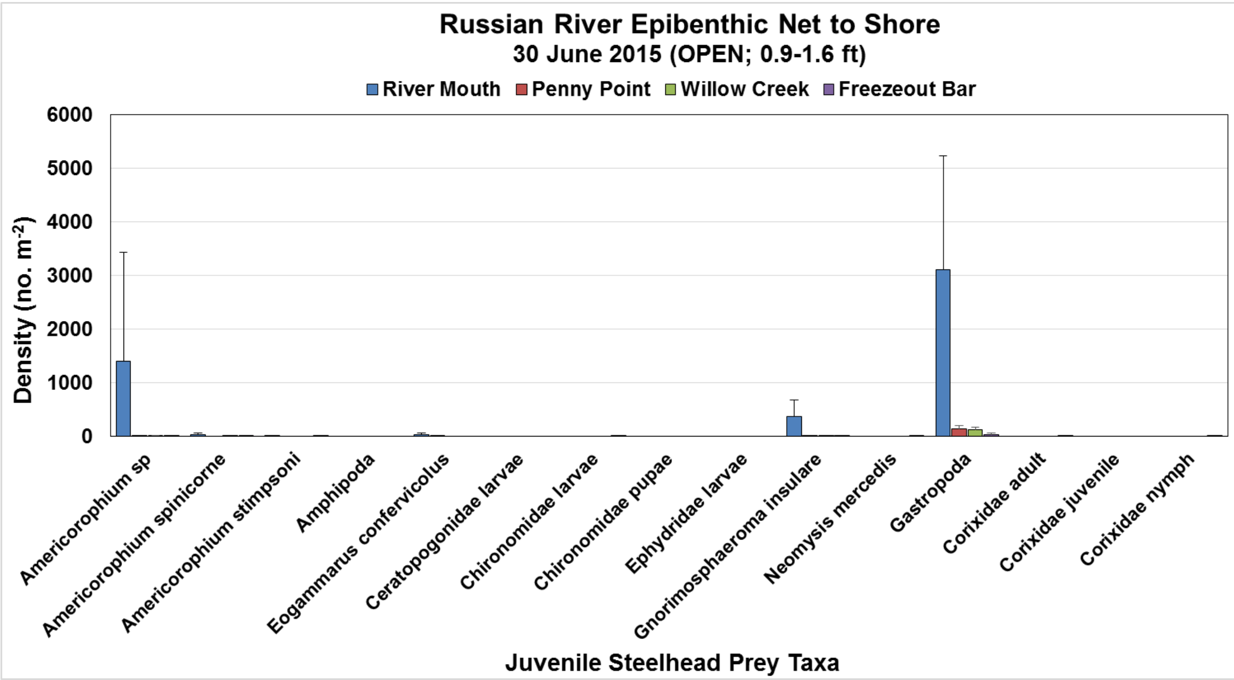


Figure 4.3.25. Density of epibenthic net macroinvertebrates documented as juvenile steelhead prey at four sites in the Russian River Estuary, 30 June 2015.

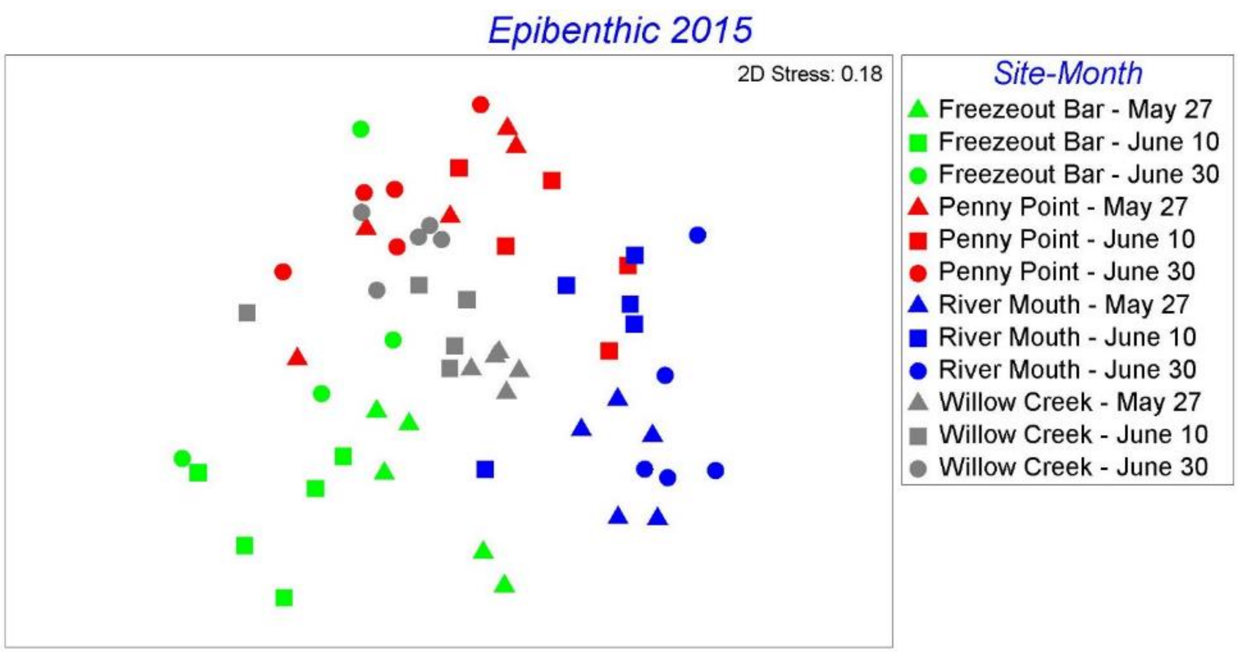


Figure 4.3 26. Multivariate analysis (NMDS) diagram of density composition of epibenthic net macroinvertebrate prey of juvenile steelhead in lower, middle and upper reaches of the Russian River Estuary, 2015.

The density composition of macroinvertebrate prey taxa in the epibenthic net samples in early June 2014 were relatively comparable to the May 27 open conditions in 2015. Most epibenthic prey were concentrated in the lower two stations, where *Americorophium* spp. and *E.*

confervicolus amphipods and *G. insulare* isopods occurred in densities as high as ~1,000 m^{-2} at River Mouth (Figure 4.3.27). Gastropod snails were also found predominantly at Willow Creek but in lower density. By late September, early into Estuary closure with the water elevation at 4.2 ft, prey taxa had diversified and expanded through the middle and upper Estuary reaches although at lower densities (Figure 4.3.28). Average densities of amphipods and isopods were ≤ 50 m^{-2} in the lower three sites but corixid beetles and dipterans—chironomid and ceratopogonid—larvae and pupae now approached up to ~100 m^{-2} in the upper reach, at Freezeout Bar. After 22 days of Estuary closure, with the water level at 6.7 ft at the Jenner Gauge, composition and densities of the same prey had expanded further into their recently inundated intertidal habitat and the aquatic insects (corixids and chironomids) averaged 114-144 m^{-2} at Freezeout Bar (Figure 4.3.29). In part, this likely represents the mobility of the epibenthic crustaceans and aquatic insects, as well as perhaps the effect of expanded, productive intertidal habitat, as compared to the benthic macroinvertebrates, which may be delayed or otherwise constrained in recruiting to the expanded habitat.

Epibenthic Sled

Samples from the epibenthic sled distinguish potential macroinvertebrate prey availability in two respects: (1) the sled samples deeper habitats parallel to the thalweg; and, (2) during prolonged closures, additional sled samples can be added where newly inundated intertidal areas are available to foraging steelhead.

Sled samples from the 27 May 2015, before Estuary closure, indicted the same general prey taxa distribution and densities as documented in the epibenthic net with the exception of increased occurrence of the mysid *Neomysis mercedis* and corixid beetles, and greater overall abundances at Freezeout Bar, in the uppermost reach (Figure 4.3.30). However, in comparison the shallower epibenthic net samples, gastropod snails were much less dense (<100 m^{-2}) in the deeper habitats (Figure 4.3.23). By June 10, in the middle of the closure, average densities were not different but the primary taxa were only dense at the River Mouth site, including gastropod snails >450 m^{-2} (Figure 4.3.31). Separation of the taxa composition and relative abundance at the routinely sampled (during open conditions) sites and the three additional, newly inundated sampling sites indicated that gastropod snails overwhelmed the abundance of the macroinvertebrates that had recently occupied the new shallow habitat, reaching levels as high as ~3200 m^{-2} and 7000 m^{-2} at Penny Point and Willow Creek, compared to ~27 m^{-2} and ~10 m^{-2}, respectively, from the deeper samples (Figure 4.3.32). Densities of most other macroinvertebrate prey were generally not different or even less dense at the River Mouth and Freezeout Bar sites. Upon return to open conditions, on 30 June the species were more uniformly distributed among the sampling sites, although *Americorophium* spp. amphipods and gastropod snails were more abundant at Freezeout Bar and *E. confervicolus* amphipods at River Mouth, but at approximately half of their relative mean densities during or before the closure. These results indicate redistributions of epibenthic and otherwise motile

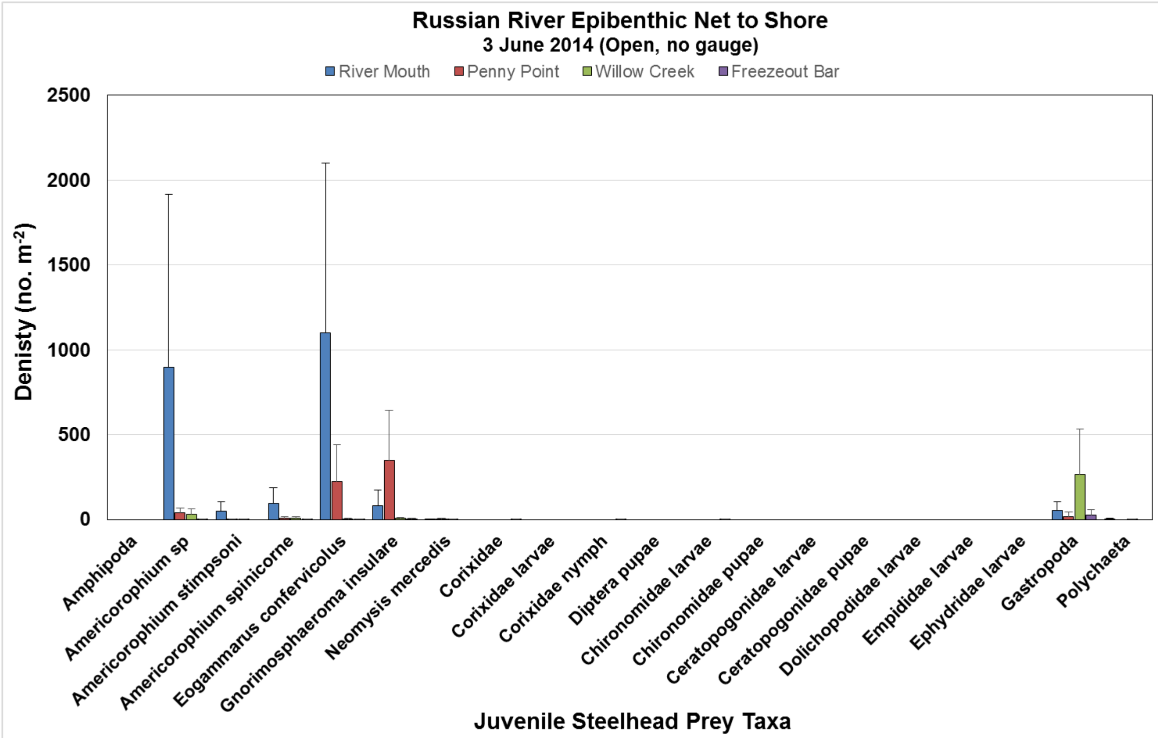


Figure 4.3.27. Density of epibenthic net macroinvertebrates documented as juvenile steelhead prey at four sites in the Russian River Estuary, 3 June 2014.

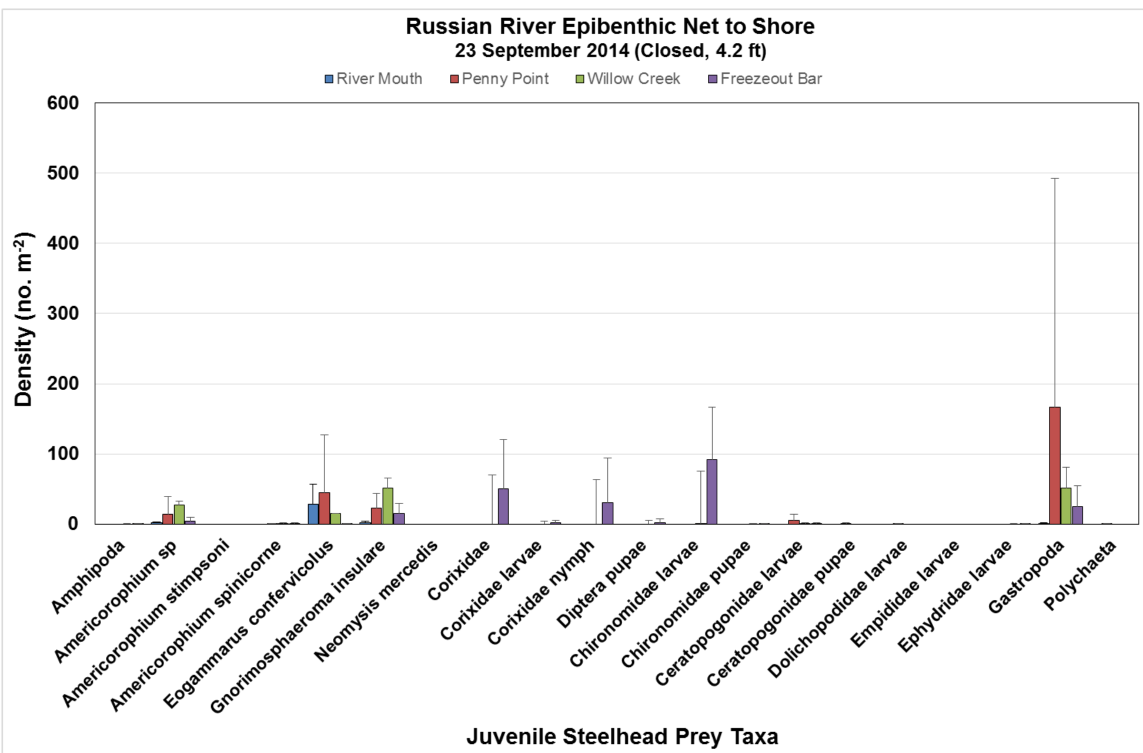


Figure 4.3.28. Density of epibenthic net macroinvertebrates documented as juvenile steelhead prey at four sites in the Russian River Estuary, 23 September 2014.

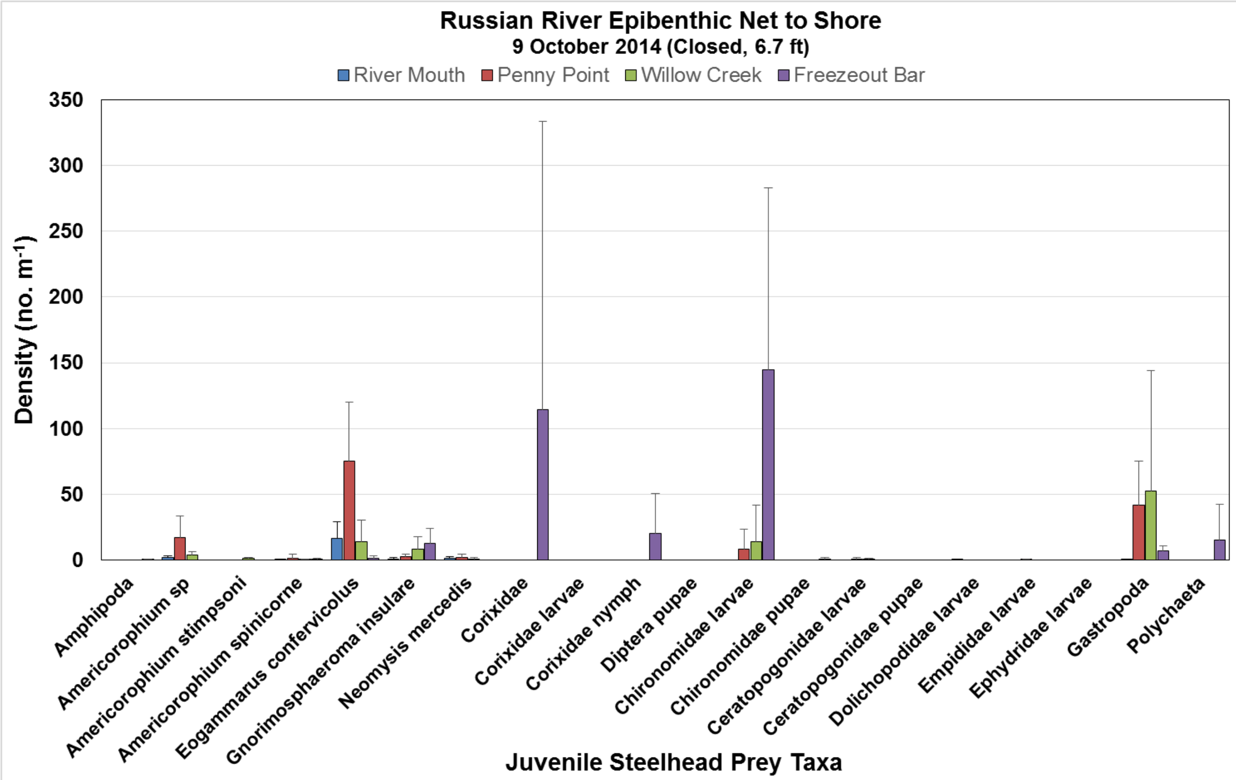


Figure 4.3.29. Density of epibenthic net macroinvertebrates documented as juvenile steelhead prey at four sites in the Russian River Estuary, 9 October 2014.

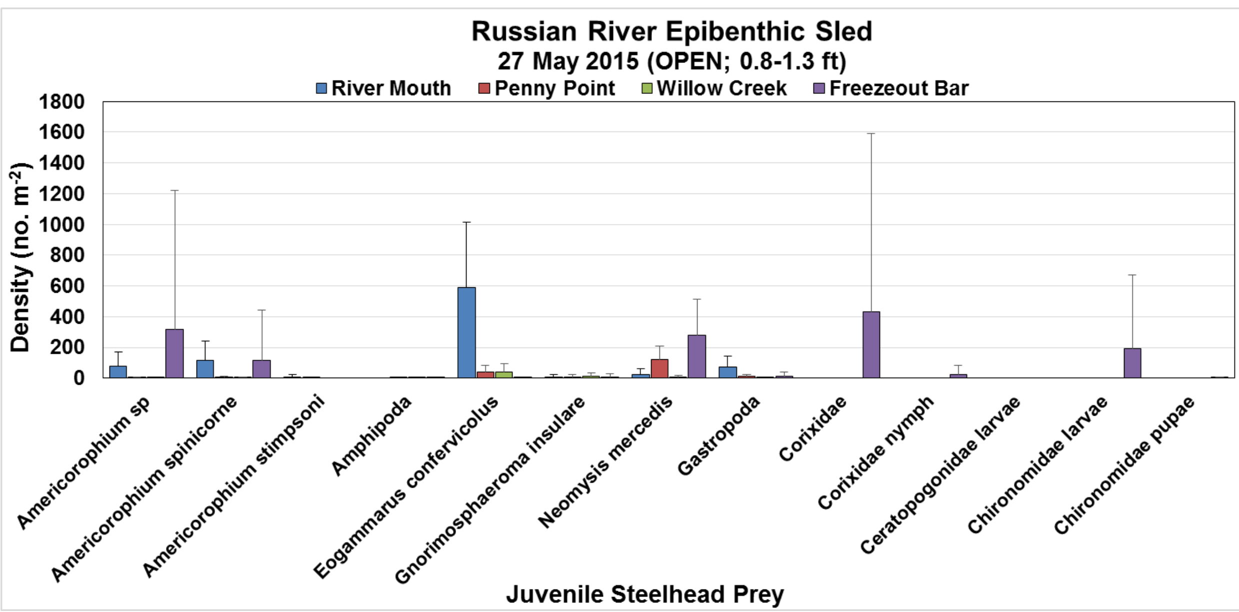


Figure 4.3.30. Density of epibenthic sled macroinvertebrates documented as juvenile steelhead prey at four sites in the Russian River Estuary, 27 May 2015.

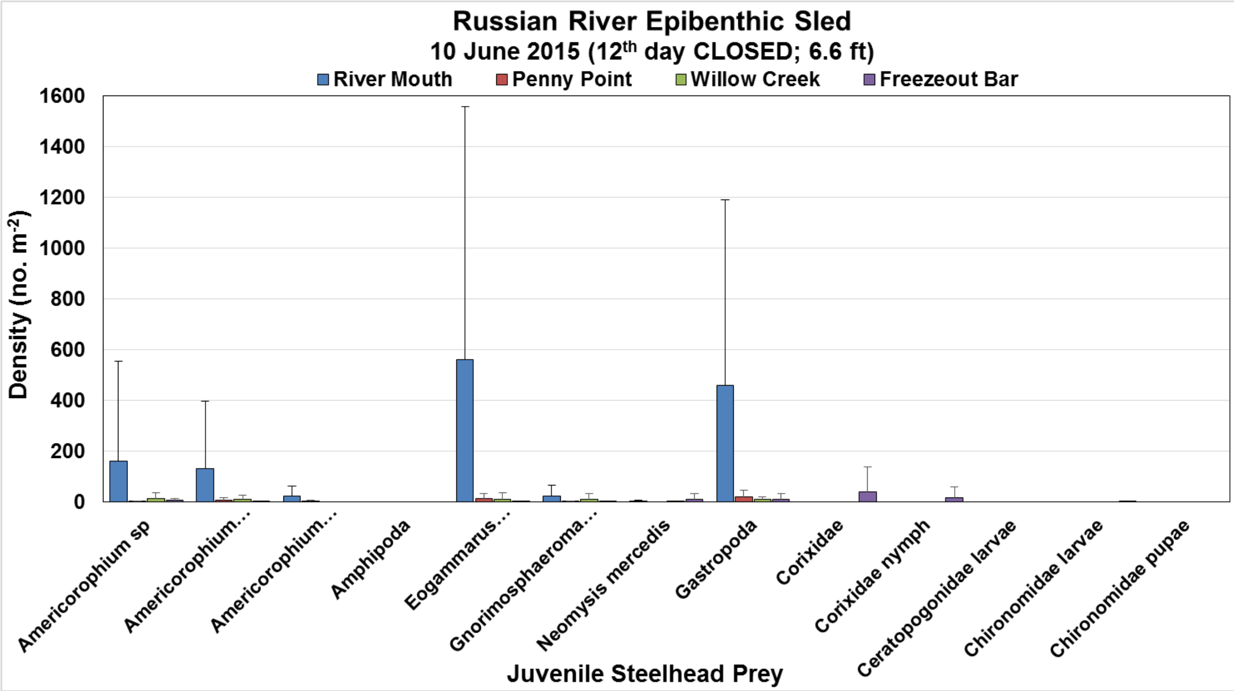


Figure 4.3.31. Density of epibenthic sled macroinvertebrates documented as juvenile steelhead prey at four sites in the Russian River Estuary, 10 June 2015; samples are the same positions as during the 27 May 2015 sampling.

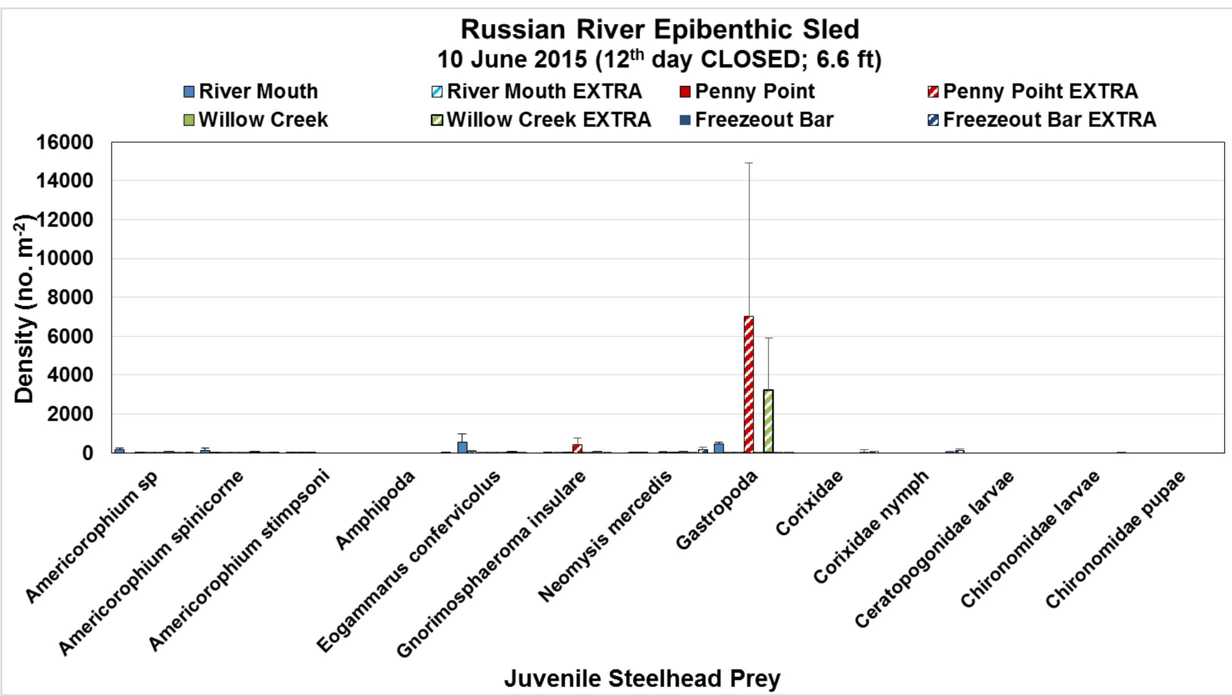


Figure 4.3.32. Density of epibenthic sled macroinvertebrates documented as juvenile steelhead prey at four sites in the Russian River Estuary, 10 June 2015; additional samples from three replicates in recently inundated intertidal zone during Estuary closure are designated by cross-hatch pattern.

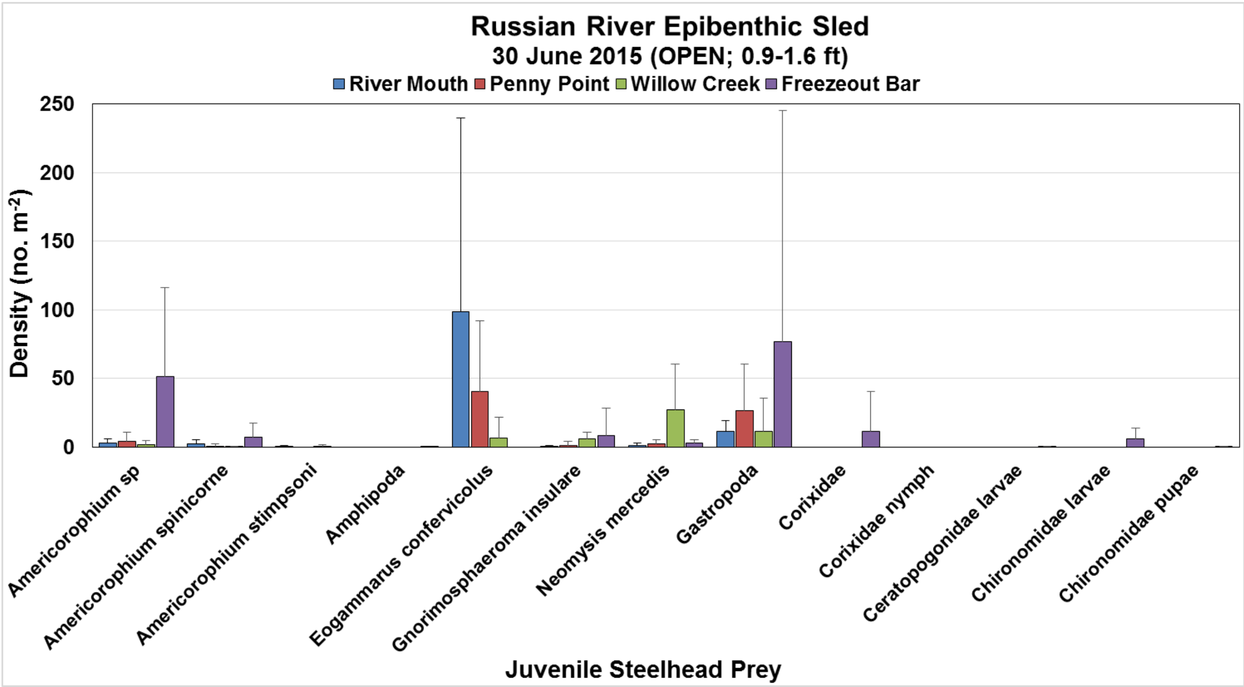


Figure 4.3.33. Density of epibenthic sled macroinvertebrates documented as juvenile steelhead prey at four sites in the Russian River Estuary, 30 June 2015.

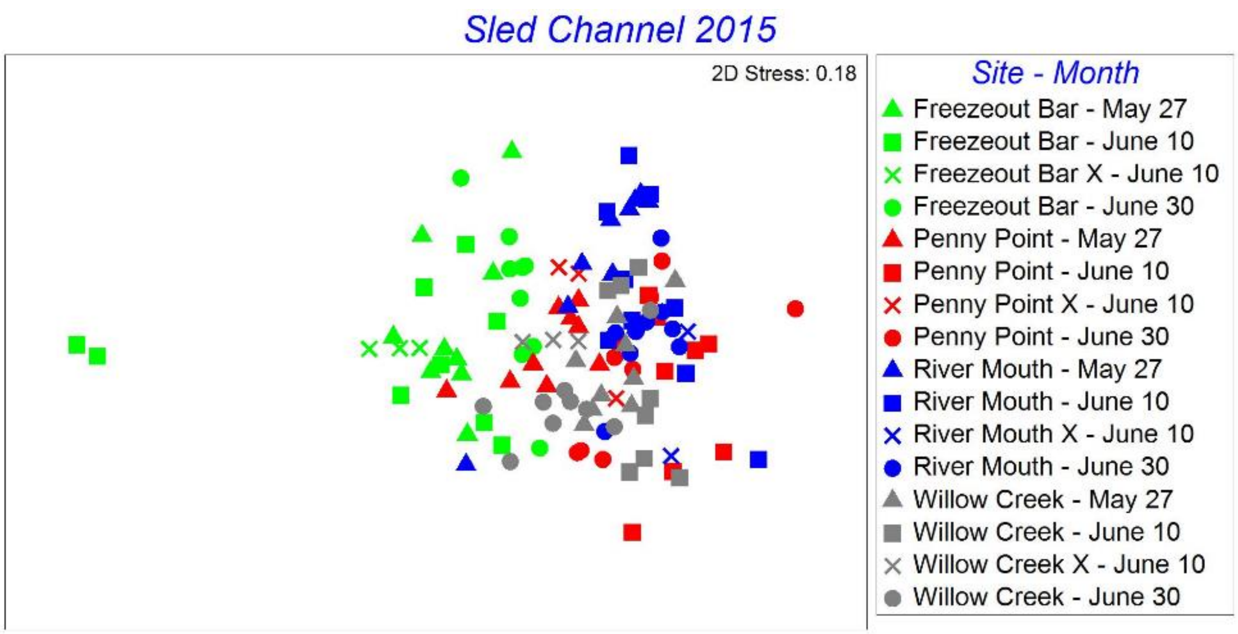


Figure 4.3.34. Multivariate analysis (NMDS) diagram of density composition of epibenthic sled macroinvertebrate prey of juvenile steelhead in lower, middle and upper reaches of the Russian River Estuary, 2015; X symbols designate samples from shallow water habitat inundated during Estuary closure.

macroinvertebrate prey into increasingly inundated shallow water habitats during Estuary closure. Decreased density of most taxa at all but the River Mouth site may indicate dispersal of relatively stable populations. Gastropod snails, however, were the only taxa that increased significantly, by over a magnitude at one site, into the newly inundated shallows.

These results from the 2015 epibenthic sled sampling reflect to some degree to those of 2014, although the temporally bracketed sampling around an estuarine closure was not available in 2014. As in 2015, the epibenthic sled samples during 2014 mirrored the epibenthic net findings with the exception of increased densities of the estuarine mysid *Neomysis mercedis* and the reduced abundance of corixid beetles. In June, there were few available prey in the upper reach at Freezeout Bar, except for higher average density (>300 m^{-2}) of mysids than in any other reach (Figure 4.3.35). Densities of other prey were on the same scale as in 2015, where *E. confervicolus* averaged ~414-515 m^{-2} at River Mouth and Penny Point, and *G. insuare* ~110 m^{-2} at Willow Creek, but *Americorophium* spp. were not dense (<50 m^{-2}) in any reach.

By September 2014 in the early stages of the late summer Estuary closure, the epibenthic amphipods, isopods and mysids and were distributed more uniformly across the Estuary but at appreciably lower densities; only *G. insulare* approached ~50 m^{-2} at Willow Creek and Freezeout Bar (Figure 4.3.36). By October, after 22 days of Estuary closure, densities of most epibenthic prey had diminished to <10 m^{-2} except for *Amercorophium* sp. and *A. spinicorne*, which had increased to 37 m^{-2} to 25 m^{-2}, respectively just at Penny Point (Figure 4.3.37). Because the positions of the two outside transects were shifted landward during the Estuary closure to compensate for the rising water elevation, the higher densities in October might suggest that the *Americorophium* spp. amphipods are moving or even increasing with shallow water inundation.

Zooplankton

In 2015, density and numerical composition of zooplankton (data filtered to remove benthic or other non-pelagic organisms and microzooplankton such as tintinnids, rotifers, and copepod nauplii) indicated lower diversity of taxa during Estuary closure on June 10 (Figure 4.3.38). Marine and estuarine plankton taxa typically dominated the lower and middle reach sites but the three downstream sites were dominated by larval gastropods and polychaetes during the closure.

Freezeout Bar had the lowest zooplankton densities before and during the Estuary closure, and was unique in the significant contribution of freshwater taxa, such as cladocerans and cyclopoid copepods, both before and during the Estuary closure. However, densities and percent composition of the oligohaline/brackish water copepod *Eurytemora affinis* were high at Freezeout Bar after the Estuary opened, likely indicating salinity intrusion into the Estuary's upper reach.

Differences in assemblage structure and abundance are evident from the multivariate analysis (Figure 4.3.39). All sites were significantly different from each other (Global R = 0.846); density composition was most similar (lower dissimilarity; 26.22 for 10 June and 31.63 for 30 June) between River Mouth and Penny Point (Table 4.3.6). Similarly, there were no significant differences among the dates before, during and after the closure (Global R = 0.924). The SIMPER analysis did indicate that taxa compositions were significantly different between dates

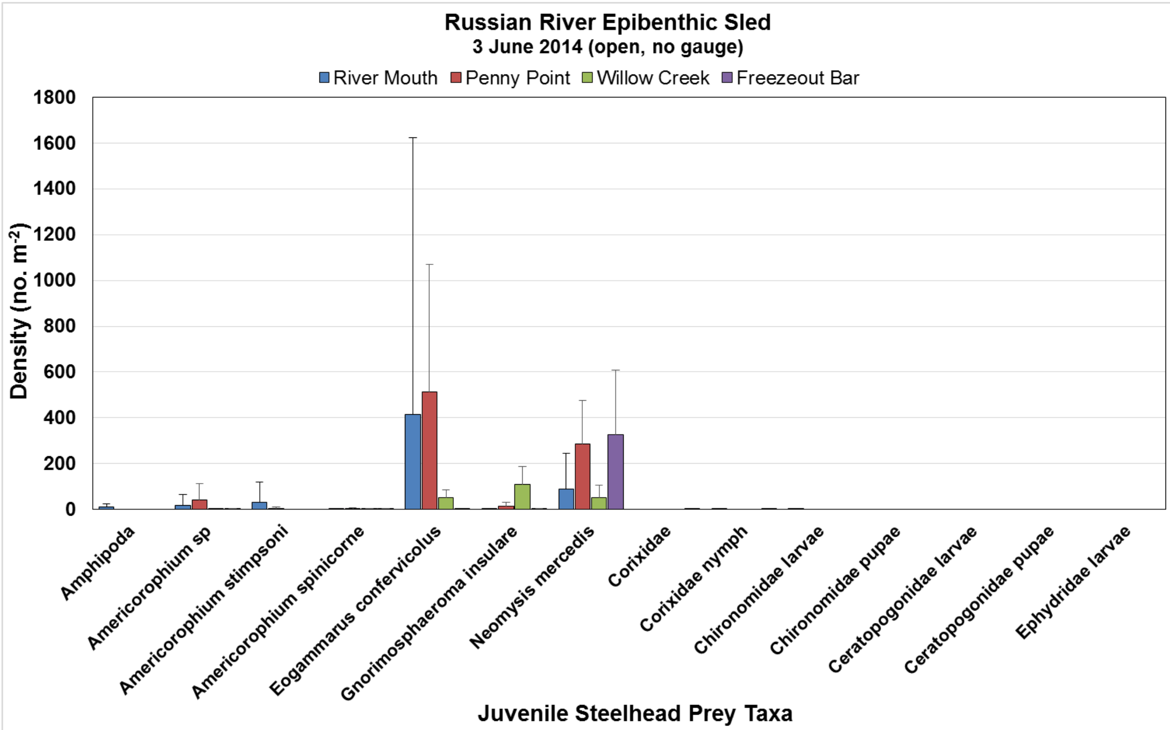


Figure 4.3.35. Density of epibenthic sled macroinvertebrates documented as juvenile steelhead prey at four sites in the Russian River Estuary, 3 June 2014.

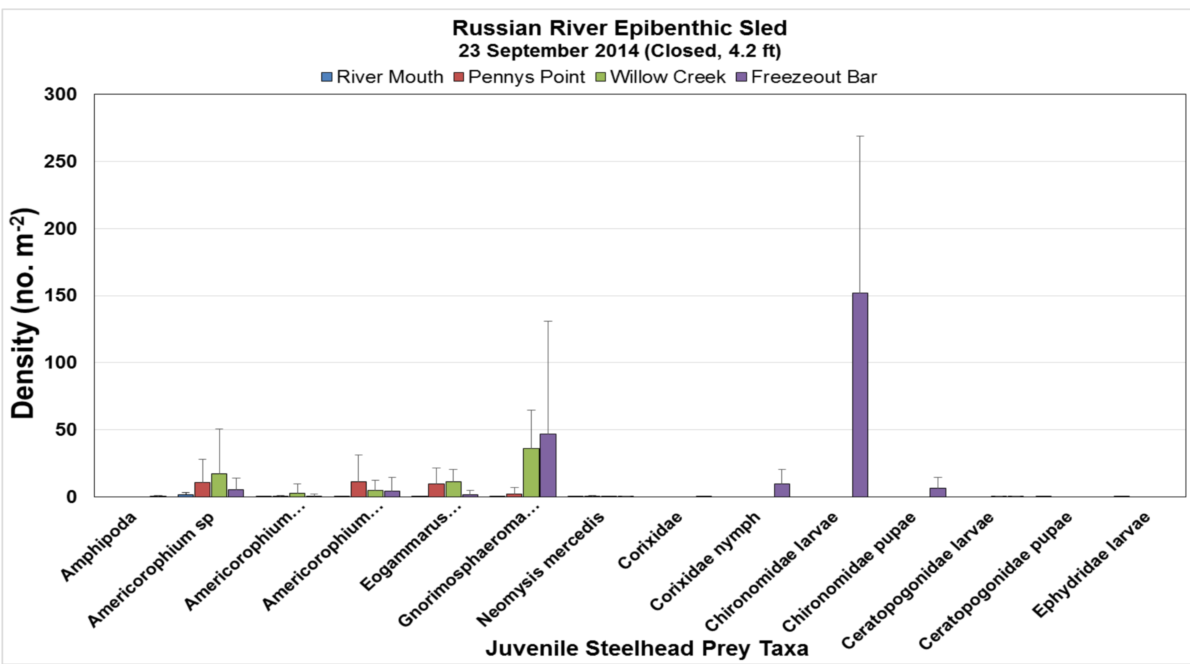


Figure 4.3.36. Density of epibenthic sled macroinvertebrates documented as juvenile steelhead prey at four sites in the Russian River Estuary, 23 September 2014.

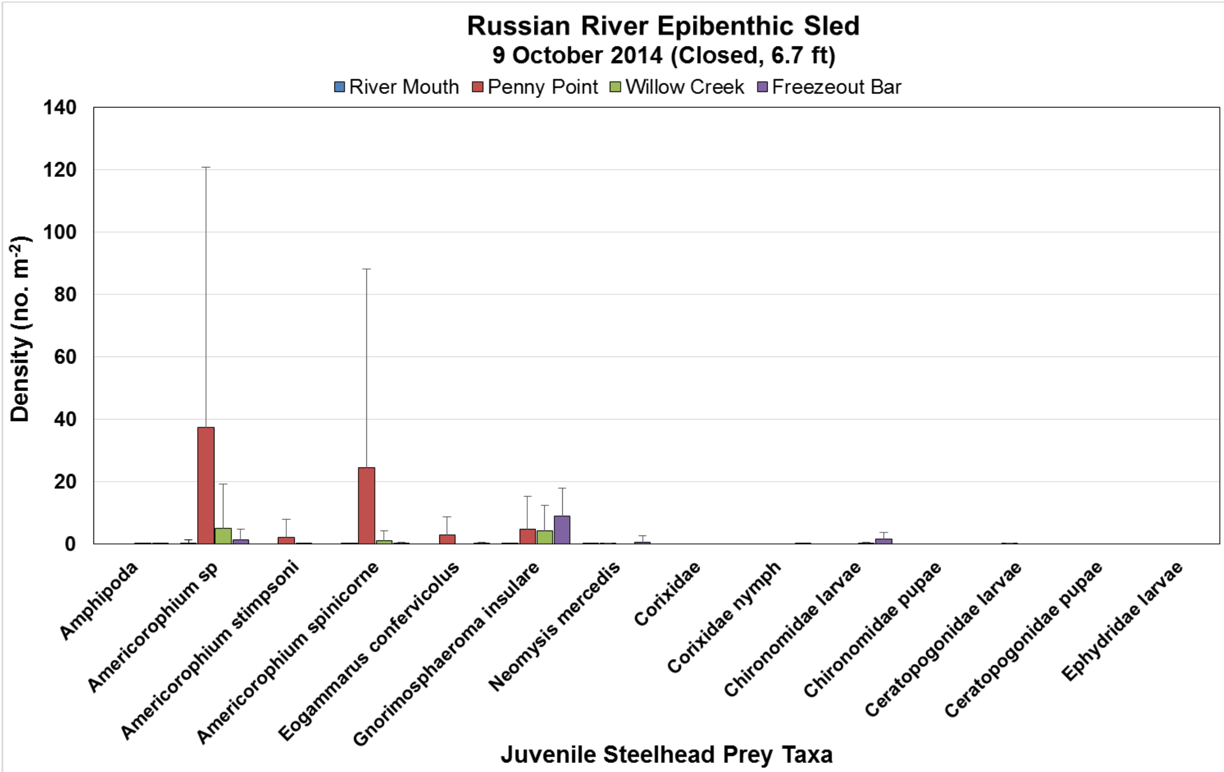
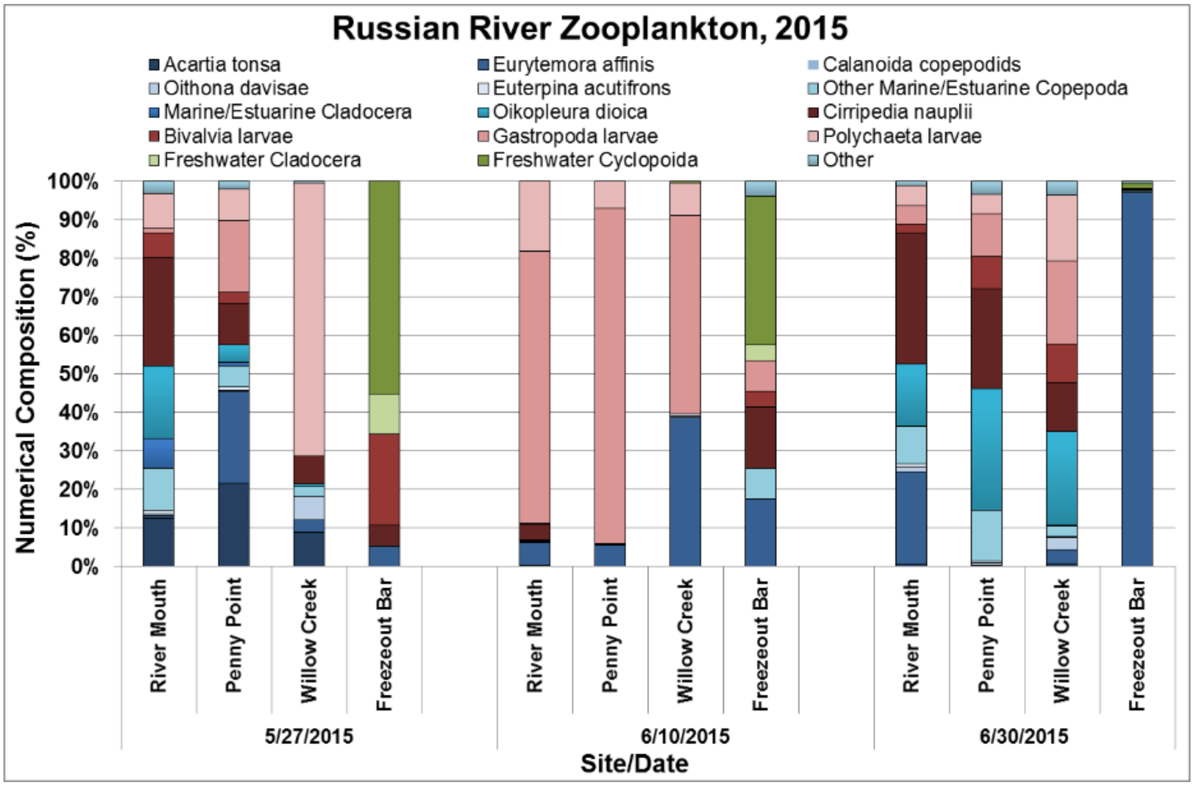


Figure 4.3.37. Density of epibenthic sled macroinvertebrates documented as juvenile steelhead prey at four sites in the Russian River Estuary, 9 October 2014.

but not among site-date combinations. The most obvious changes in taxa density composition occurred between 27 May and 10 June at Willow Creek (dissimilarity = 82.51) and 10 June and 30 June at Freezeout Bar (80.29) (Table 4.3.6).

In 2014, density and numerical composition of zooplankton (filtered to remove benthic or other non-pelagic organisms and microzooplankton such as tintinnids, rotifers, and copepod nauplii, as for 2015) indicated highest densities at Willow Creek and Penny Point in June, during open-Estuary conditions (Figure 4.3.40). Among the sites, Freezeout Bar had relatively low zooplankton densities both during the June open Estuary and during the Estuary closures in September and October. As in 2015, the Freezeout Bar site was unique in the significant contribution of freshwater taxa, such as cladocerans and cyclopoid copepods, particularly during the Estuary closure on 23 September.



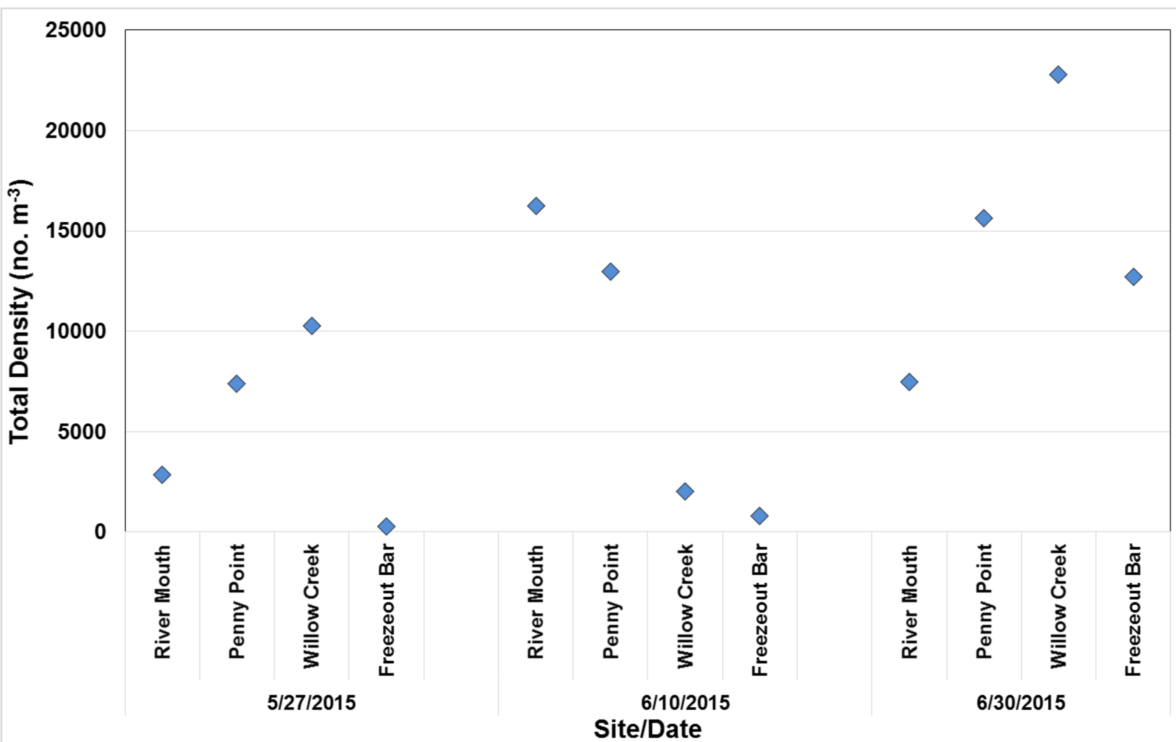


Figure 4.3.38. Zooplankton percent composition (top) and total densities (bottom) before, during and after 2015 Estuary closure; tintinnids, rotifers, and copepod nauplii have been removed from the data.

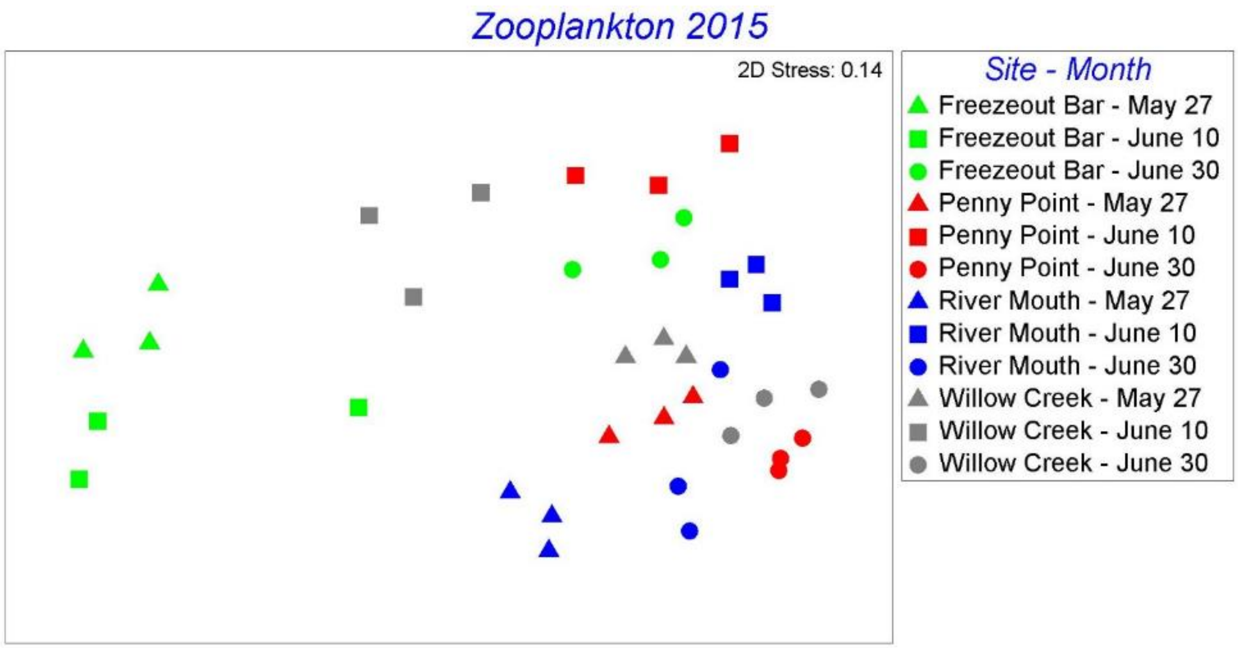
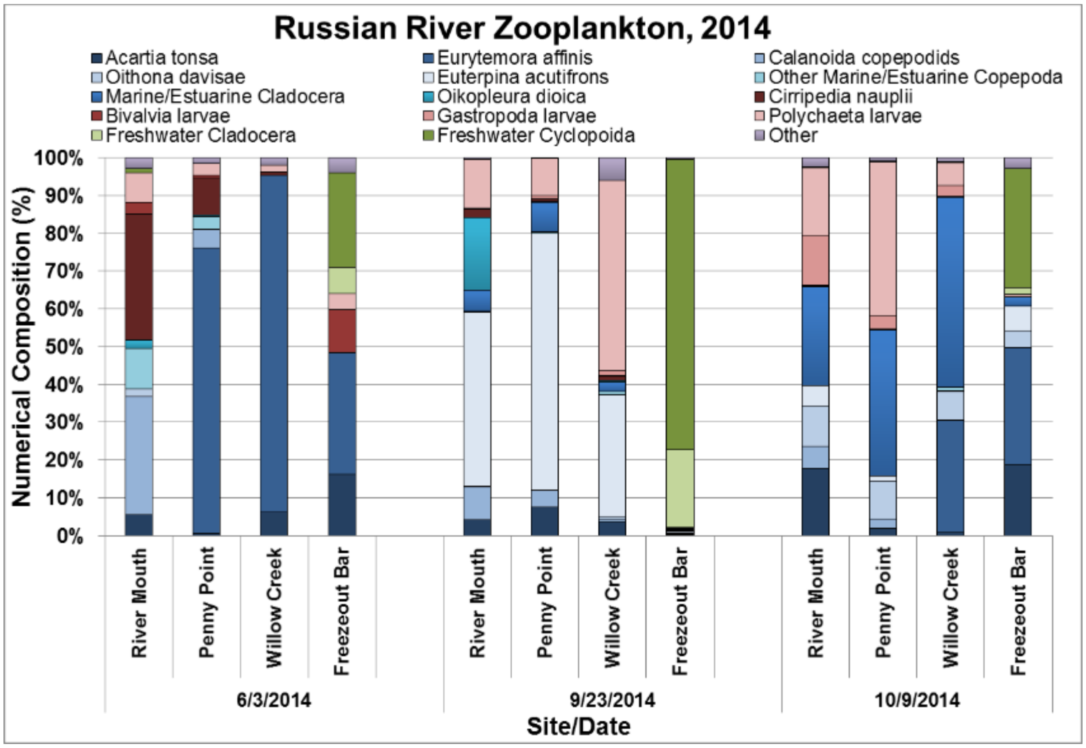


Figure 4.3.39. Multivariate analysis (NMDS) diagram of selected zooplankton assemblages at four sites on three dates in the Russian River Estuary, 2015.

Table 4.3.6. Bray-Curtis dissimilarity values of zooplankton numerical composition from four sites on three dates before, during and after closure of Russian River Estuary, 2015.

		River Mouth			Penny Point			Willow Creek			Freezeout Bar		
		27-May	10-Jun	30-Jun	27-May	10-Jun	30-Jun	27-May	10-Jun	30-Jun	27-May	10-Jun	30-Jun
River Mouth	27-May		70.27	52.82	41.4	48.8	46.4	58.28	88.16	48.91	94.15	90.73	93.7
	10-Jun			59.02	82.57	26.22	71.65	67.81	78.97	58.96	97.57	95.21	77.96
	30-Jun				64.36	61.09	31.63	68.76	86.19	38.41	96.37	95.32	87.52
Penny Point	27-May					59.25	55.74	61.6	73.8	44.85	96.07	93.57	75.46
	10-Jun						71.5	67.82	75.48	65.84	97.82	94.74	74.69
	30-Jun							76.11	91.75	44.5	97.92	97.68	96.13
Willow Creek	27-May								82.51	54.59	95.71	90.77	90.68
	10-Jun									77.34	96.04	81.71	66.29
	30-Jun										95.09	92.74	85.02
Freezeout Bar	27-May											67.01	89.95
	10-Jun												80.29
	30-Jun												



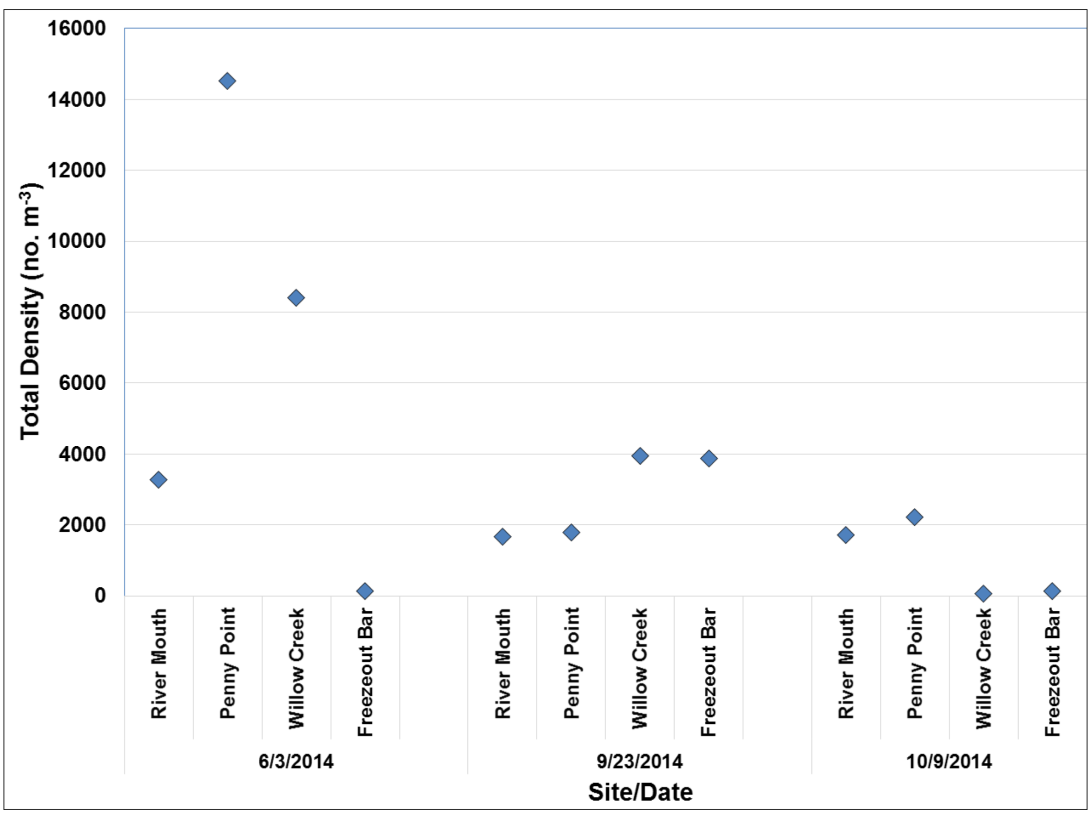


Figure 4.3.40. Zooplankton percent composition (top) and total densities (bottom) before and during Russian River Estuary closures in 2014; tintinnids, rotifers, and copepod nauplii have been removed from the data.

Marine and estuarine plankton taxa dominated the lower and middle reach sites, most notably *Acartia tonsa*, *Eurytemora affinis* and other calanoid copepods or cladocerans, as well as the neritic harpacticoid copepod *Euterpina acutifrons*. These marine/estuarine plankters did appear, albeit in low densities, at Freezeout Bar in June and October, suggesting salinity intrusion into the Estuary's upper reach. The copepod *E. affinis* appears to be a definite indicator of oligohaline/brackish water bodies because it did not occur at River Mouth at any time but was particularly prominent at Penny Point and Willow Creek in June, and a prominent component of the plankton assemblage at those sites during the long closure in October.

This difference in assemblage structure and abundance among the sampling sites is readily evident from the multivariate analysis (Figure 4.3.41). While the density composition appears to be most similar among River Mouth and Penny Point, and to a lesser extent Willow Creek, plankton is always comparatively distinct due to the more distinct Freezeout Bar assemblages; Global R is high for differences among site groups (Global R = 0.91) and date groups (Global R = 1). As described above, *E. affinis* was the dominant contributor to the similarity in plankton assemblage structure at Willow Creek (75.5%) and Penny Point (0.52%) in June and Freezeout Bar in October (26.7%).

Summary

Findings

Relationship of Epibenthic Prey Availability to Juvenile Steelhead Diet

As demonstrated in diet composition documented through this study since 2009, juvenile steelhead occupying the Estuary tend to feed somewhat specifically on a limited suite of epibenthic crustaceans and aquatic insects. These prey are dominated by two species of gammarid amphipods, tube-dwelling *Americorophium* spp. (*A. spinicorne*; *A. stimpsoni*) and epibenthic *Eogammarus confervicolus*, the epibenthic isopod *Gnorimosphaeroma insulare*, mysid *Neomysis mercedis,* and aquatic insects of the hemipteran family Corixidae (water boatmen) also occur consistently in the diets of juvenile steelhead sampled in other estuaries along the northeastern Pacific, including other intermittent systems (Needham 1940; Shapovalov and Taft 1954; Meyer *et al.* 1981; Martin 1995; Salamunovich and Ridenhour 1990; Daly *et al.* 2014). Only in a few cases, of small, persistent estuarine lagoons such a Waddell Creek, have other prey such as aquatic insects become more prominent (Needham 1940).

This dominantly epibenthic feeding strategy indicates that juvenile steelhead in this, and seemingly most estuaries, are foraging along the bottom, whether in deeper channel or shallower, marginal habitats. The only deviation from this comparatively consistent prey spectrum is the inclusion of gastropod snails that appeared much more prominently in the diets of juvenile steelhead from 2015 collections in the Estuary. This species has been tentatively identified as the non-indigenous, euryhaline New Zealand mudsnail, *Potmopyrgus antipodarum,* which have also been shown to occur in juvenile Chinook salmon diets (Bersine *et al.* 2008). They occurred primarily in diets of juvenile steelhead in the upper reach of the Estuary in late summer, but also occurred in high density in epibenthic sled samples from the recently

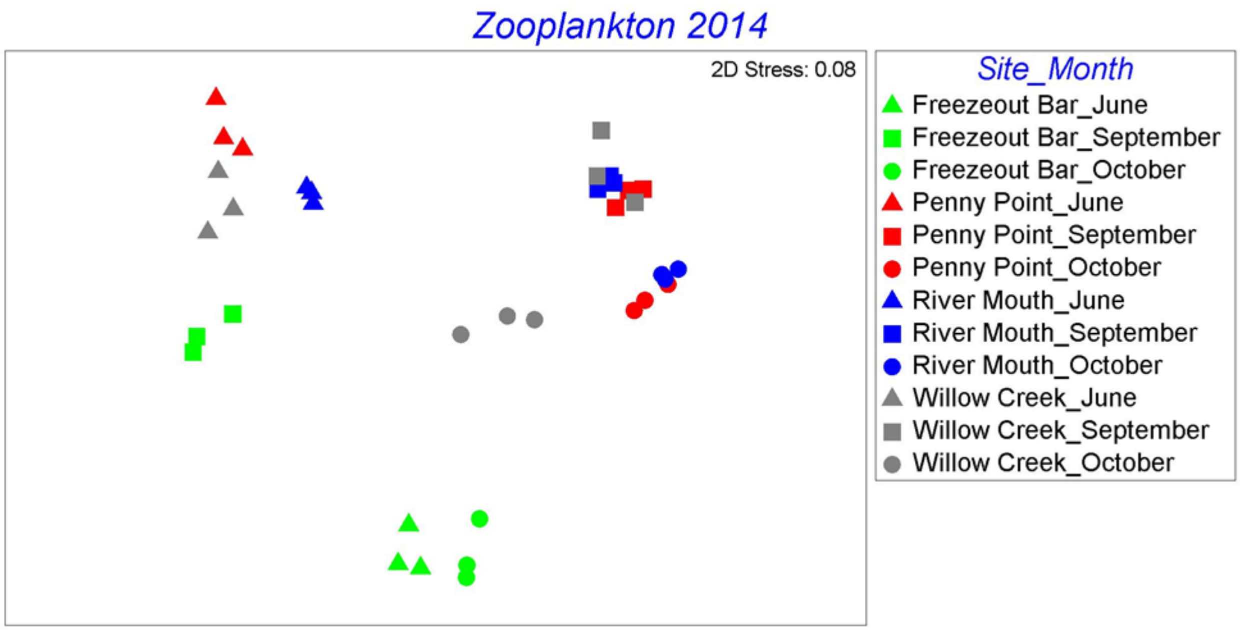


Figure 4.3.41. Multivariate analysis (NMDS) diagram of selected zooplankton assemblages at four sites on three dates in the Russian River Estuary, 2014.

inundated shallow water on 10 June, suggesting that the snails are sufficiently mobile to occupy the new habitat within a short period of time; they are ovoviviparous and brood their young, so populations can expand locally independent of water flow or other hydrologic influences. We are uncertain whether the occurrence of these snails in the diets of juvenile steelhead indicates selection for these benthic invertebrates or their overwhelming density in the shallow water habitat in which juvenile steelhead are foraging.

Conversely, except for an occasional larval or juvenile fish, essentially none of the prominent pelagic taxa in the zooplankton samples occur in the documented juvenile steelhead diets. Despite seemingly viable prey, such as the calanoid copepods *Acartia* spp. and *Eurytemora affinis*, occurring in densities as high as 400-1,500 m^{-3} in our zooplankton sampling, they have not appeared to any degree in the diets of steelhead in the Estuary from 2009-2016. The only exceptions among the zooplankton samples as well as the epibenthic samples are the relatively rare insects that appear in juvenile steelhead diets. Other than the corixid beetles, that are often fed upon prominently by juvenile steelhead, especially in the upper reach of the Estuary, insect adults, pupae and larvae could be fed upon the sediment surface (larvae, pupae), in the water column as they emerge in pupation, and as adults drifting on the sediment surface.

Other than the evident foraging orientation toward epibenthic invertebrates, these results do not necessarily reflect high prey selectivity by juvenile steelhead in the Estuary because the apparent low diversity of available prey taxa in this intermittent Estuary generally mirrors the steelhead prey spectrum. Other than the relatively rare cases of truly benthic fauna—the gastropod snails and nereid polychaetes annelids (which are also known to swim up into the water column)—there are very few abundant taxa in the epibenthic net or sled samples that are not prominent in the documented steelhead diets. This would suggest that prey availability is somewhat well characterized by the Water Agency-UW/WET epibenthic sampling in the Estuary. Results of the benthic sampling also reflects many of the diet components, in part because some taxa such as the *Americorophium* spp. amphipods build tubes in the sediment but also because epibenthic forms on the sediment surface are also captured by the benthic core sampling method. However, there is an extensive array of abundant benthic infauna taxa in the core samples that seldom or never appear in juvenile steelhead diets in the Estuary. For instance, taxa of Foraminifera, Nematoda, Oligochaeta are enumerated in densities of up to 10,000's to 100,000's m^{-2} in these samples.

Prey availability varies naturally over time and space under open Estuary conditions. In general, densities of prey organisms are higher early in the sampling period and diminish by roughly an order of magnitude by late summer. Some of the major prey taxa also occur in the highest densities in the lower reach early in the season but their distribution eventually expands into the middle and upper reach, potentially related to the expansion of oligohaline conditions and stratification. In the 2015 epibenthic net sampling, the highest densities of epibenthic amphipods and isopods occurred predominantly at River Mouth in late May and remained so when the Estuary was open again in late June; gastropod snails appeared prominently at Willow Creek. Conversely, the epibenthic sled sampling indicated more equivalent densities throughout the Estuary, especially at Freezeout Bar in late May, and somewhat uniformly available at all sites after the Estuary opened in late June even though they were highly concentrated only at the River Mouth site during the closure.

In 2014, closed Estuary conditions from late September to early October, most of the epibenthic amphipods and isopods were equally or more dense in the middle and upper reaches than the lower reach, and aquatic insects (larvae and pupae, as well as adult corixids) dominated the prey assemblage in the upper reach, at Freezeout Bar. The mysid *Neomysis mercedis* was the only potential prey that occurs somewhat uniquely, being present in relatively high abundance in the deeper portions of the channel at all sites early in the study season. We have observed them to appear in dense patches during Estuary closures, which would suggest that our sampling may not accurately characterize their occurrence and availability to juvenile steelhead.

Responses of Prey Availability to Estuary Closure

Prey composition and densities from the epibenthic net and channel sled samples were relatively comparable in both 2015 and 2014, suggesting that there was equal or a relatively minor gradient of prey density distribution from their deeper channel to shallower marginal habitats. The prominent exception were corixids, which in 2014 occurred almost exclusively in the epibenthic net to shore samples in the upper reach, suggesting that they were available only in shallow water within 10 m of the shoreline. Coincidentally, it should be noted that, unlike other years of this study, in 2014 the corixids did not appear prominently in steelhead diet. In 2015, corixids occurred in 25% of the fish examined although they did not contribute materially to the numerical or gravimetric proportion of all prey consumed; however, they were prominent (mean density ~434 m^{-2}) in the epibenthic sled samples at Freezeout Bar in late May, and persisted at slightly lower densities throughout and after the June closure.

Prey availability sampling bracketing the 16-day closure in 2015 offers a more comparable measure of the response of juvenile steelhead prey to the effects of an Estuary closure. Epibenthic net sampling indicated that amphipod (prominently *E. confervicolus*), isopod and gastropod snail prey moved into the recently inundated shallow intertidal habitats that was available by 10 June; this was most notable for the River Mouth site in the case of *E. confervicolus* and Penny Point and Willow Creek for the snails. *E. confervicolus* were equally or slightly more dense (mean ~118 vs. 65 m^{-2} density) at this shallow water edge during the closure than before. This redistribution into the increased shallow water habitat with the closure was also evident in the findings from the epibenthic sled at the River Mouth, wherein *E. confervicolus* amphipods remained at relatively the same densities at the same sampling sites before and during the closure, and the additional samples available because of new shallow water inundation suggested their movement to the increasing shallow water habitat. The occurrence of gastropod snails was even more indicative of movement and potential concentration in shallower water, wherein mean densities of ~3200-7000 m^{-2} were found in the new shallow sampling sites at Penny Point and Willow Creek as compared to only 10-20 m^{-2} at the repeated transect sampling at the same sites, and ~500 m^{-2} at the River Mouth site.

Variation in Zooplankton Density Composition

Zooplankton assemblage composition and densities varied consistently, particularly in the contrast between the lower three sites occupied by marine and estuarine taxa, and the upper, Freezeout Bar, site occupied more by oligohaline and typical tidal freshwater taxa. High abundance of the estuarine copepod *Eurytemora affinis* at Freezeout Bar when the Estuary opened after closure in 2015 may indicate that they were able to colonize there during the closure and because it was such a dry year, they were able to stay there after opening—note that they were never that abundant at Freezeout in 2014.

It may be worth noting that taxa with weak swimming abilities (harpacticoids and gastropod and polychaete larvae in 2015; the harpacticoid copepod *E. acutifrons* in 2014) experienced increased abundances and contributions to percent composition during Estuary closures. However, the time between sampling events alone could have accounted for these differences, especially for 2014.

Recommendations

Findings from juvenile steelhead diet and prey availability from 2009 through 2015, and especially through the definitive 2015 Estuary closure period, augment several recommendations for an approach to reorient continued monitoring and research in the Estuary to refine our understanding of the implications of Estuary management for juvenile steelhead and other salmonids. A separate document being prepared for submission to a scientific journal will present the results of juvenile steelhead behavior based on hydroacoustic tagging and other related investigations during 2014-2015 (Matsubu *et al*. In prep.).

Shifting to Prey-Specific Processing of Prey Availability Samples

Strategic modification of the protocols for laboratory processing of prey availability samples should be considered to improve relevance and completeness of that task. The high diversity and often high density of macroinvertebrates in the benthic, epibenthic net and sled and zooplankton samples requires considerable time devoted to identifying and enumerating these samples in the UW/WET laboratory. This typically requires establishing sample processing priorities that prevent many of the samples being processed. This strategy enables the most relevant samples (e.g., bracketing an Estuary closure) being processed, but other spatial and temporal patterns of potential consequence and interest left incomplete. Given the extremely consistent prey selection by juvenile salmon, which has been established in these studies since 2009 (Seghesio 2011), the project could appreciably increase the efficacy of the documentation of prey availability by selectively processing the epibenthic net and sled samples to the ~14-20 taxa that reflect known or likely prey, rather than the entire spectrum of macroinvertebrate taxa. Presently, considerable laboratory processing time and expertise is allocated to enumerating taxa (e.g., ostracods, nematodes, oligochaetes, foraminiferans, turbellarians) that occur rarely, if at all, in juvenile steelhead diets. While the total biotic community dataset is unusually complete and valuable in its own right, it is now sufficiently documented to consider such a strategic change, which would increase the likelihood that all samples in any field season could be processed for the target juvenile steelhead prey availability. Furthermore, sampling would retain all organisms and these would be archived at the UW/WET and available at a later date for additional processing if required. Similarly, for zooplankton, there are few recognized prey of juvenile steelhead in the samples, and revising the processing protocol to avoid identifying and counting the numerous benthic or other non-pelagic organisms and microzooplankton such as tintinnids, rotifers, and copepod nauplii would result in much more relevant characterization of those taxa in the Estuary's zooplankton assemblage that respond to Estuary closures. An alternative would be to process all taxa during Estuary closure periods; benthic samples might also be considered a separate case, in terms of the multiple uses that dataset provides.

Demography and Production of Prey Populations in Response to Estuary Closure

Despite revisions in the Water Agency and UW/WET study design and sampling protocols that are more adaptive to assessing changes in prey availability with Estuary closure, there is still considerable uncertainty about both the effects of Estuary closure on prey populations and the ability of juvenile steelhead to exploit them. As we have refined our understanding of the natural variability in patterns of juvenile steelhead foraging and prey availability over space and time in the normally open Estuary, future monitoring and research should consider concentrated investigations of responses to Estuary closures. This could logically involve two stages: (1) continued processing and analysis of epibenthic net and sled prey availability samples that remain unprocessed; and, (2) dedicated, "pulse" field investigations during future periods of Estuary closure. The purpose of this deeper delving into prey availability would be to address the present uncertainty about the source and consequence of epibenthic prey immigrating or otherwise occupying shallow intertidal habitat with increasing water elevations after the Estuary closes. Specific questions would investigate hypotheses such as: (a) extant epibenthic prey populations volitionally expand and disperse into the increasing areas of shallow water habitat; (b) production of epibenthic organisms increases as a function of increased availability of organic detritus and other food resources; (c) predation pressure from foraging by juvenile steelhead on epibenthic prey increases with inundation of shallow intertidal habitat; and, (d) rapid decrease in water elevation after re-opening of the closed Estuary imposes mortality to epibenthic prey populations in occupied shallow water habitats. Such hypotheses can be addressed in part by existing samples and data, and future Water Agency sampling, but will also require supplemental sampling and experiments, preferably encompassing one or more future closure events, at much greater sampling frequency and intensity.

Enhanced Steelhead Diet and Foraging Rate Data Collection

Differences in potential consumption rate, indicated by patterns in the size-specific instantaneous ration in prior years and in both 2015 and 2014, imply potential reach and Estuary status differences in availability among the suite of preferred prey taxa. While the instantaneous ration is a viable index of consumption rate (e.g., Figure 4.3.15), consideration should be given to conducting periodic diet sampling of juvenile steelhead over a 24-hr or 30-hr period in order to obtain a more precise estimate of daily ration, which is a fundamental measurement for bioenergetic modeling of potential growth. It should be recognized that this involves periodic sampling during nocturnal hours, which may be unfeasible given Water Agency policies or resources.

Tracking Distribution and Role of New Zealand Mud Snail in Juvenile Steelhead Diet and Prey Availability

Given our 2009-2015 database of benthic and epibenthic macroinvertebrates, and the relatively recent outbreak of New Zealand mud snails in both juvenile steelhead diets and prey availability samples in the upper reach of the Estuary, it would be worthwhile to initiate further analysis of this invasion. An obvious approach we would already plan is to ensure that the bioenergetic value as prey for juvenile steelhead will be evaluated in the bioenergetic modeling. Furthermore, tracking relative change in prey selection between the snail and co-occurring native epibenthic amphipods, isopods and insects would be a natural enhancement of the *Demography and Production of Prey Populations in Response to Estuary Closure* studies. Assessing the survival of the snails under dewatering of the shallow intertidal when the Estuary reopens after a long closure would also be informative from the standpoint of Estuary management.

4.4 Fish Sampling – Beach Seining

The Water Agency has been fish sampling the Russian River Estuary since 2004 - prior to issuance of the Biological Opinion. An Estuary fish survey methods study was completed in 2003 (Cook 2004). To provide context to data collected in 2015, we present and discuss previous years of data in this report. Although survey techniques have been similar since 2004, some survey locations and the sampling extensity changed in 2010 as required in the Biological Opinion. The distribution and abundance of fish in the Estuary are summarized below. In addition to steelhead, coho salmon, and Chinook salmon, we describe the catch of several common species to help characterize conditions in the Estuary.

Methods

Study Area

The Estuary fisheries monitoring area included the tidally-influenced section of the Russian River and extended from the sandbar at the Pacific Ocean to Duncans Mills, located 9.8 km (6.1 mi) upstream from the coast (Figure 4.4.1).

Fish Sampling

A beach-deployed seine was used to sample fish species, including salmonids, and determine their relative abundances and distributions within the Estuary. The rectangular seine consisted of 5 mm (¼ inch) mesh netting with pull ropes attached to the four corners. Floats on the top and weights on the bottom positioned the net vertically in the water. From 2004 to 2006, a 30 m (100 ft) long by 3 m (10 ft) deep purse seine was used. From 2007 to 2014 a conventional seine 46 m (150 ft) long by 4 m (14 ft) deep was used. Then in 2015 a 46 m by 3 m seine with a 3 m square pocket located in the center of the net was employed. The seine was deployed with a boat to pull an end offshore and then around in a half-circle while the other end was held onshore. The net was then hauled onshore by hand. Fish were placed in aerated buckets for sorting, identification, and counting prior to release.

Salmonids were anesthetized with Alka-Seltzer tablets or MS-222 and then measured, weighed, and examined for general condition, including life stage (i.e., parr, smolt). All salmonids were scanned for passive integrated transponder (PIT) tags or other marks. Steelhead and coho salmon were identified as wild or hatchery stock by a clipped adipose fin. Hatchery coho salmon were no longer clipped after spring 2013 and were either marked with a coded wire tag or PIT tag. Tissue and scale samples were collected from some steelhead. Unmarked juvenile steelhead caught in the Estuary greater than 60 mm fork length were surgically implanted with a PIT tag. Fish were allowed to recover in aerated buckets prior to release.

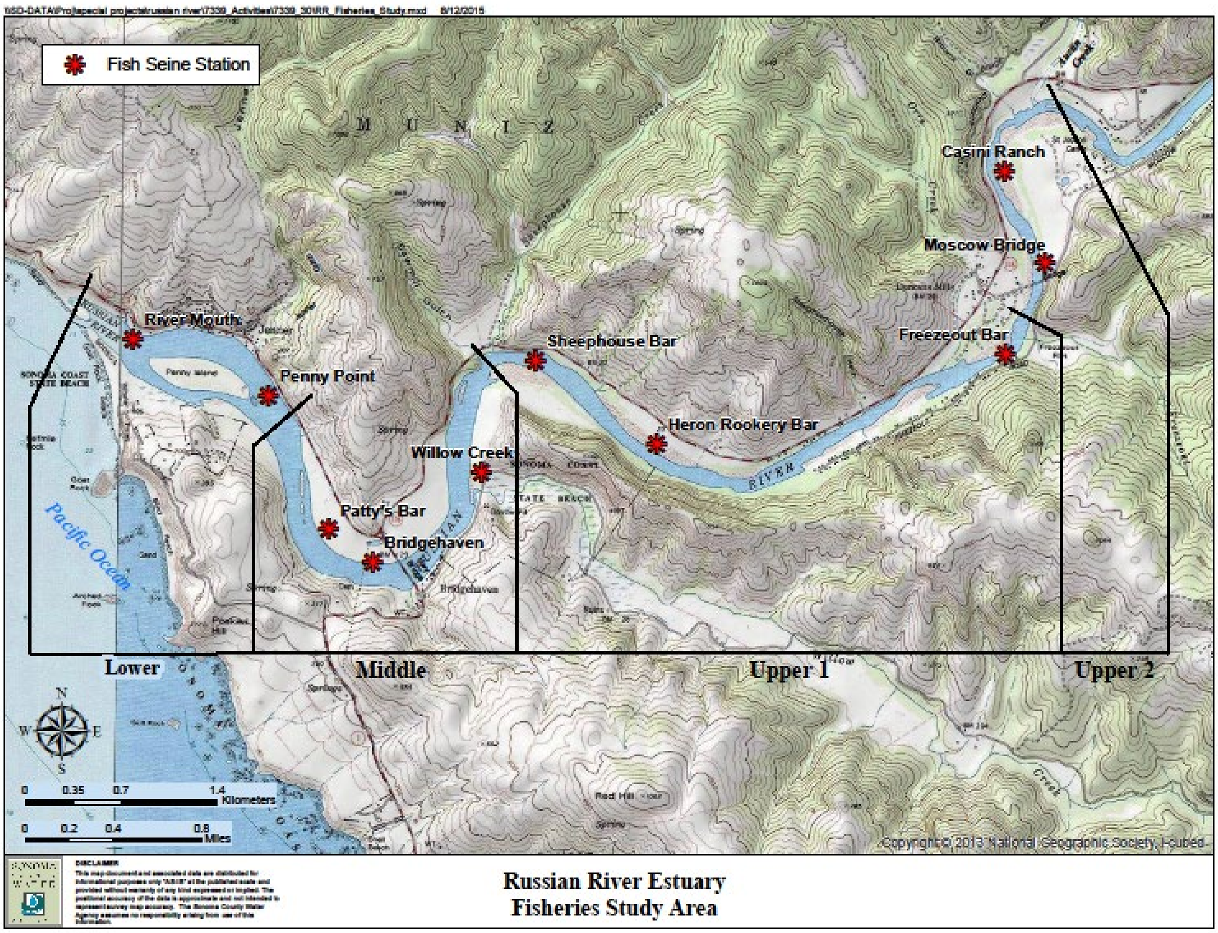


Figure 4.4.1. Russian River Estuary fisheries seining study reaches and sample sites, 2015.

From 2004 to 2009, eight seining stations were located throughout the Estuary in a variety of habitats based on substrate type (i.e., mud, sand, and gravel), depth, tidal, and creek tributary influences. Three seine sets adjacent to each other were deployed at each station totaling 24 seine sets per sampling event. Stations were surveyed approximately every 3 weeks from late May through September or October. Total annual seine pulls ranged from 96 to 168 sets.

Starting in 2010 fish seining sampling was doubled in effort with 300 sets completed for the season. Surveys were conducted monthly from May to October. Between 3 and 7 seine sets where deployed at 10 stations for a total of 50 sets for each sampling event. Twenty-five sets were in the lower and middle Estuary and 25 in the upper Estuary. In 2014 and 2015 the seining sampling effort was conducted in May, June, and September to characterize the Estuary under tidal conditions during the beginning and end of the lagoon management period. In 2014 seining was also conducted in October. Seining in July and August were not completed because a lagoon outlet channel could not be installed to form a freshwater lagoon.

For data analysis the Estuary study area was divided into three reaches, including Lower, Middle, and Upper, which is consistent with study areas for water quality and invertebrate studies (Figure 4.1.1). For the fish seining study, the Upper Reach of the Estuary was divided into Upper1 and Upper2 sub-reaches to improve clarity on fish patterns. Fish seining stations were located in areas that could be sampled during open and closed river mouth conditions. Suitable seining sites are limited during closed mouth conditions due to flooded shorelines. Catch per unit effort (CPUE), defined as the number of fish captured per seine set (fish/set), was used to compare the relative abundance of fish among Estuary reaches and study years.

The habitat characteristics and locations of study reaches, fish seining stations, and number of monthly seining sets are below:

- Lower Estuary
    - River Mouth (7 seine sets): sandbar separating the Russian River from the Pacific Ocean, sandy substrate with a low to steep slope, high tidal influence.
    - Penny Point (3 seine sets): shallow water with a mud and gravel substrate, high tidal influence.
- Middle Estuary
    - Patty's Bar (3 seine sets): large gravel and sand bar with moderate slope, moderate tidal influence.
    - Bridgehaven (7 seine sets): large gravel and sand bar with moderate to steep slope, moderate tidal influence.
    - Willow Creek (5 seine sets): shallow waters near the confluence with Willow Creek, gravel and mud substrate, aquatic vegetation common, moderate tidal influence.
- Upper Estuary

    Upper1 Sub-Reach
    - Sheephouse Bar (5 seine sets): opposite shore from Sheephouse Creek, large bar with gravel substrate and moderate to steep slope, low to moderate tidal influence
    - Heron Rookery Bar (5 seine sets): gravel bank adjacent to deep water, low to moderate tidal influence.
    - Freezeout Bar (5 seine sets): opposite shore from Freezeout Creek, gravel substrate with a moderate slope, low tidal influence.

    Upper2 Sub-Reach
    - Moscow Bridge (5 seine sets): steep to moderate gravel/sand/mud bank adjacent to shallow to deep water, aquatic vegetation common, low tidal influence.
    - Casini Ranch (5 seine sets): moderate slope gravel/sand bank adjacent to shallow to deep water, upper end of Estuary at riffle, very low tidal influence.

Results

Fish Distribution and Abundance

Fish captures from seine surveys in the Russian River Estuary for 2014 are summarized in Table 4.4.1. During the 12 years of study 50 fish species were caught in the Estuary. In 2015, seine captures consisted of 29,227 fish comprised of 26 species. No new fish species were detected in the Estuary during 2015 fish seining.

The distribution of fish in the Estuary is, in part, based on a species preference for or tolerance to salinity (Figure 4.4.2). In general, the influence of cold seawater from the ocean under open mouth conditions results in high salinity levels and cool temperatures in the Lower Reach transitioning to warmer freshwater in the Upper Reach from river inflows (Figure 4.4.3). The water column is usually stratified with freshwater flowing over the denser seawater.

Fish commonly found in the Lower Reach were marine and estuarine species including topsmelt (*Atherinops affinis*), surf smelt (*Hypomesus pretiosus*), and staghorn sculpin (*Leptocottus armatus*). The Middle Reach had a broad range of salinities and a diversity of fish tolerant of these conditions. Common fish in the Middle Reach included those found in the Lower Reach and shiner surfperch (*Cymatogaster aggregata*) and bay pipefish (*Syngnathus leptorhynchus*). Freshwater dependent species, such as the Sacramento sucker (*Catostomus occidentalis*), Sacramento pikeminnow (*Ptychocheilus grandis*), and Russian River tule perch (*Hysterocarpus traskii pomo*), were predominantly distributed in the Upper Reach. Anadromous fish, such as steelhead (*Oncorhynchus mykiss*) and American shad (*Alosa sapidissima*), which can tolerate a broad range of salinities, occurred throughout the Estuary. Habitat generalists, such as threespine stickleback (*Gasterosteus aculeatus*) and prickly sculpin (*Cottus asper*), occurred in abundance in the Estuary, except within full strength seawater in the Lower Reach.

Estuary water conditions changed during a river mouth closure from September 7 to October 4, 2015, which disconnected the Estuary from tidal circulation and flooded shoreline areas with fresh river flows (Figure 4.4.3). Salinity levels at the surface decreased in the Lower Estuary while a wedge of bottom brackish water migrated upstream into the Upper Estuary. Water temperatures became more uniform throughout the Estuary due to the backwater effect of warmer river flows, which increased surface temperatures in the Lower Estuary.

There was a substantial change in the distribution of fish groups when the river mouth closed and formed a lagoon (Figure 4.4.2). Under tidal conditions in June 2015 the fish composition was dominated by estuarine and generalist species in the Lower and Middle Reaches and generalists and freshwater species in the Upper Reach. Under closed mouth conditions in September, which increased freshwater throughout the surface of the Estuary, there was a broader separation of the fish groups. Estuarine species were more abundant in the Lower Estuary, while freshwater species were more abundant in the Upper2 Reach. Also, generalist species shifter from Upper1 Reach to the Middle Reach.

Table 4.4.1. Total fish caught by beach seine in the Russian River Estuary, 2015. Each station was sampled monthly during May, June, and September for a total of 150 seine sets for all sites. Monthly seine sets per station are shown in parentheses.

		Seining Station										
Life History	**Species**	**River Mouth (7)**	**Penny Point (3)**	**Patty's Bar (3)**	**Bridge-haven (7)**	**Willow Creek (5)**	**Sheep-house Bar (5)**	**Heron Rookery Bar (6)**	**Freeze-out Bar (4)**	**Moscow Bridge (5)**	**Casini Ranch (5)**	**Total**
Anadromous	American shad									9	141	150
	Chinook salmon	11	1		67	27	42	42		3	1	194
	coho salmon	34	1	6	31	4	5	25				106
	steelhead			1	3	3	8	4	7	12	12	50
Estuarine	bay pipefish	4	2		4	3	1					14
	shiner surfperch		16	10	33	81	52	53				245
	staghorn sculpin	113	89	22	19	35	21	29	2			330
	starry flounder	65	44	6	1	39	10	19	8		36	228
	topsmelt	568	110	131	196	41						1046
Freshwater	black crappie											
	bluegill											
	California roach								3	19	56	78
	common carp											
	cyprinid sp								1			1
	fathead minnow											
	green sunfish											
	hardhead											
	hitch								40	73		113
	largemouth bass								13	12	4	29
	mosquitofish											
	Russian River tule perch					2	1	66	209	451	291	1020
	Sacramento blackfish											
	Sacramento pikeminnow					6	1	21	3	283	50	364
	Sacramento sucker		3	31	98	121	491	2535	548	774	391	4992
	white catfish									1		1
Marine	buffalo sculpin											
	cabezon	25										25
	English sole											
	northern anchovy	1			2							3

Life History	Species	Seining Station: River Mouth (7)	Penny Point (3)	Patty's Bar (3)	Bridge-haven (7)	Willow Creek (5)	Sheep-house Bar (5)	Heron Rookery Bar (6)	Freeze-out Bar (4)	Moscow Bridge (5)	Casini Ranch (5)	Total
	Pacific herring	11										11
	Pacific sanddab	81										81
	poacher sp.											
	saddleback gunnel											
	Sebastes sp.	149	3									152
	sharpnose sculpin											
	shortnosed sculpin											
	silver spotted sculpin											
	surf smelt				7							7
	jacksmelt											
	kelp greenling											
	lingcod											
	Pacific sand sole											
	Pacific sardine											
	penpoint gunnel											
	smoothead sculpin											
	snailfish sp											
	striped kelpfish											
	tidepool sculpin											
	arrow goby					1						1
Generalist	prickly sculpin*	23	151	184	145	102	48	31	7	29	7	727
	threespine stickleback	10	1423	1588	1191	2921	1976	8521	453	998	178	19259
Grand Total		**1136**	**1845**	**1986**	**1863**	**3419**	**2700**	**11406**	**1341**	**2682**	**1180**	**29227**

*Prickly Sculpin counts may include small numbers of the freshwater-resident Coast Range sculpin (*Cottus aleuticus*) and riffle sculpin (*Cottus gulosus*), although neither of these species has been reported from the Estuary.

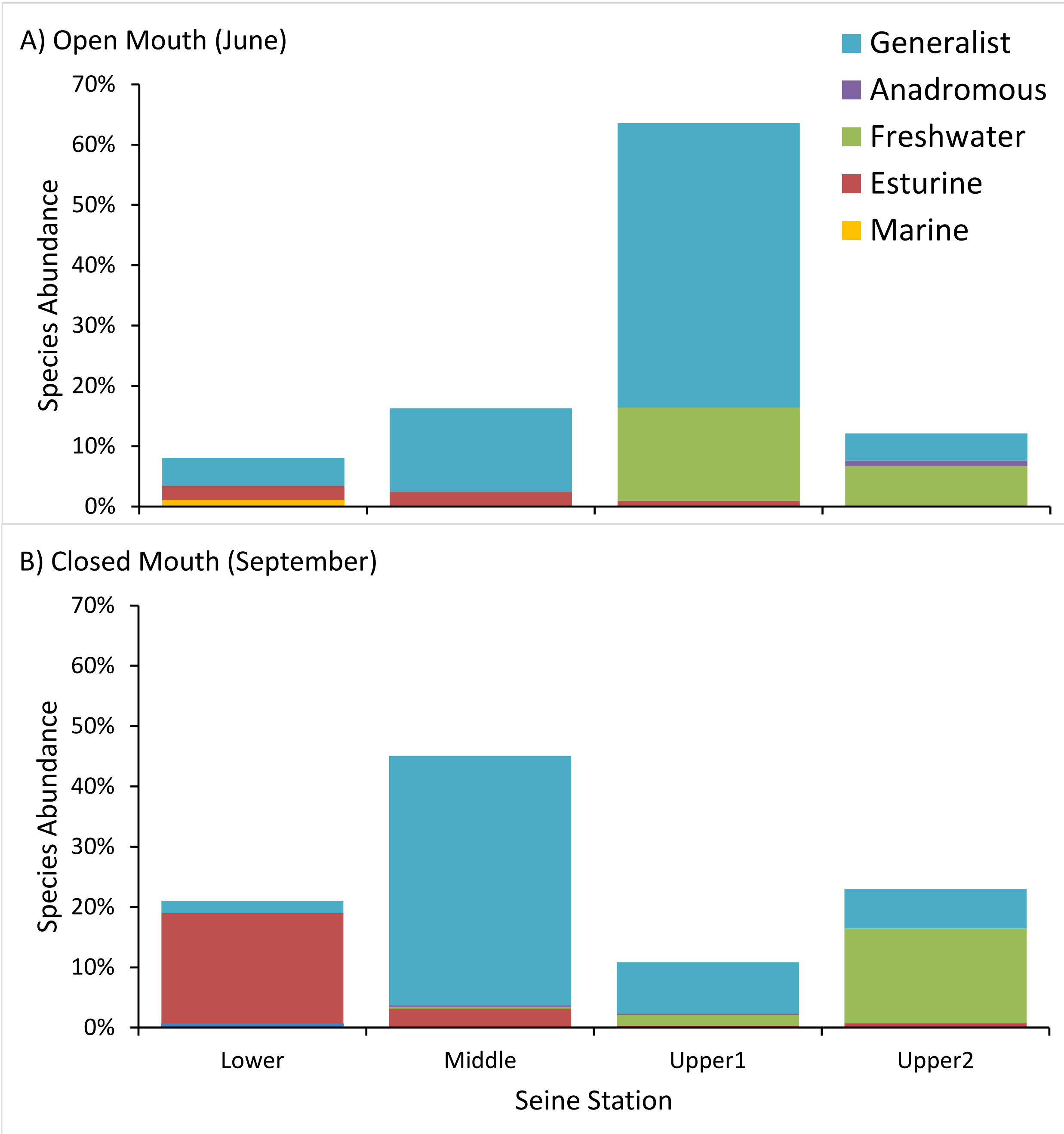


Figure 4.4.2. Distribution of fish in the Russian River Estuary based on salinity tolerance and life history, 2015. Data is from monthly seining during May, June, and September. Panel A) tidally influenced Estuary in June. Panel B) closed mouth lagoon in September. Groups include: generalist species that occur in a broad range of habitats; species that are primarily anadromous; freshwater resident species; brackish-tolerant species that complete their lifecycle in estuaries; and species that are predominantly marine residents. See Table 4.4.1 for a list of species in each group.

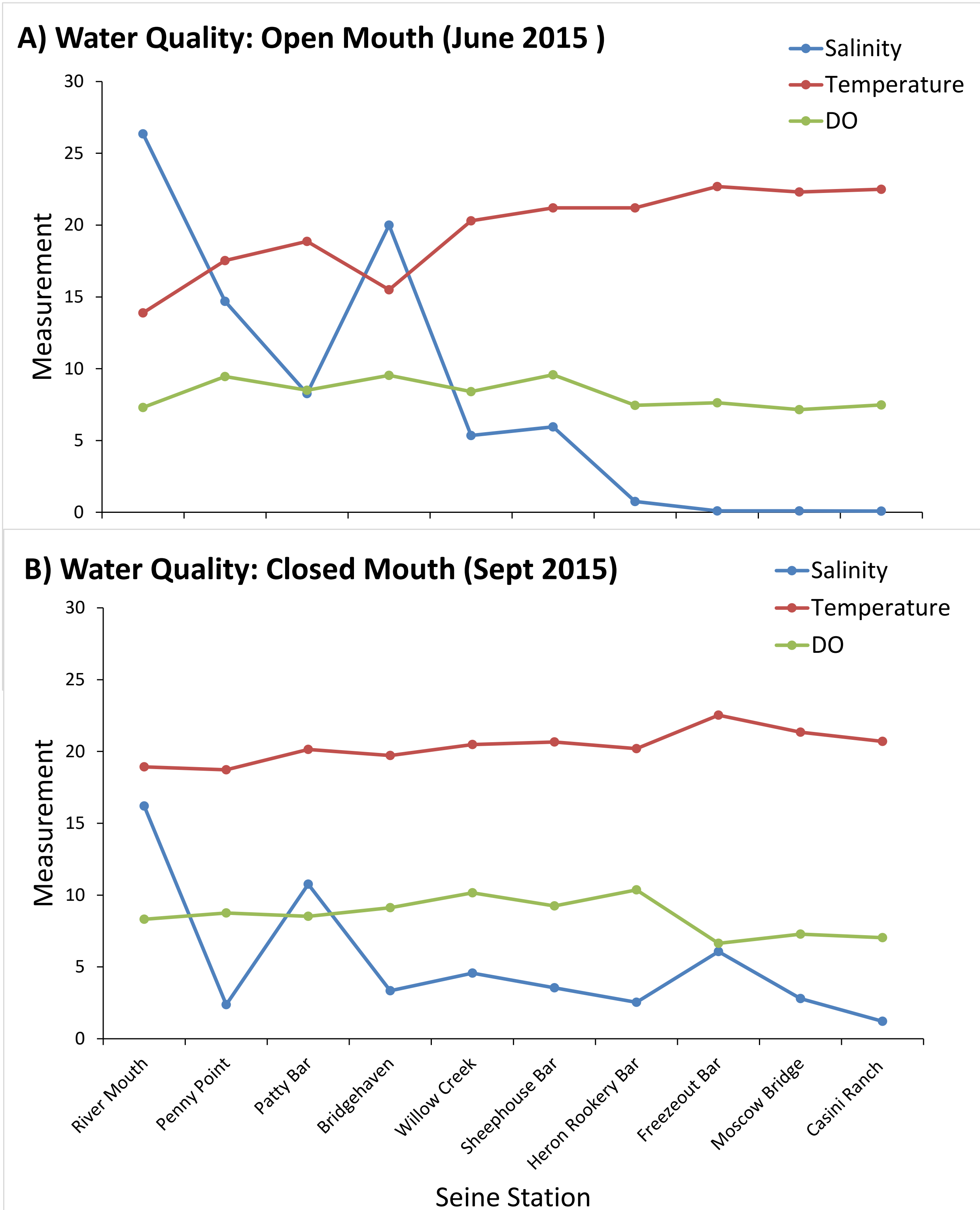


Figure 4.4.3. Generalized water quality conditions at fish seining stations in the Russian River Estuary, 2015. Values are averages collected at 0.5 m intervals in the water column during beach seining events from A) June during open river mouth conditions and B) September during closed mouth conditions. Salinity values are in parts per thousand (ppt), dissolved oxygen (DO) milligrams per liter (mg/L), and water temperature Celsius (C).

Steelhead

During 2015, a total of 50 steelhead were captured (Table 4.4.1) in 150 seine sets. The resulting CPUE was 0.67 fish/set (Figure 4.4.4). In comparison, during 2014, a total of 56 steelhead were captured in 150 seine sets for a CPUE of 0.28 fish/set. The highest CPUE for all study years was 1.66 fish/set in 2008. All steelhead captured in 2015 were wild. The seasonal abundance of steelhead captures varied annually in the Estuary (Figure 4.4.5). Juvenile steelhead were captured during all three survey events in 2015. The highest steelhead abundances are typically in June and August. During 2015, steelhead captures were highest during May at 1.08 fish/set. The highest capture abundance among all study years was in August at 4.3 fish/set and June at 4.2 fish/set in 2008. Since seining surveys began in 2004, steelhead appear to have a patchy distribution and vary in abundance in the Estuary (Figure 4.4.6). Over all years surveyed, captures were typically highest in the Upper Reach with a high of 6.9 fish/set in the Upper1 Sub-Reach in 2008.

The temporal and spatial distribution of juvenile steelhead in the Estuary in 2015 was strongly influenced by relatively large captures in the Upper1 and Upper2 in May and June (Figure 4.4.7). A few late season steelhead were caught in Upper2 and Middle Reaches. No steelhead were captured in the Lower Reach. However, several steelhead were seine-captured at Jenner Gulch (Lower Estuary) for a telemetry study.

Most captured juvenile steelhead were age 0+ parr or age 1+ smolts and ranged in size from 55 mm to 368 mm fork length (Figure 4.4.8). Estuary steelhead in May appeared to consist of age 0+ parr less than 100 mm fork length, age 1+ smolts up to 200 mm fork length, and few large steelhead greater than 288 mm. Steelhead parr and smolts in September ranged in size from 105 mm to 368 mm fork length.

In 2015, 87 juvenile steelhead captured during Estuary seining surveys and a telemetry study conducted by the University of Washington were implanted with PIT tags. Also, 1892 juvenile steelhead where PIT-tagged during downstream migrant trapping studies in the Russian River and tributaries upstream of the Estuary. There were two smolt steelhead tagged in the Estuary and recaptured. One fish was tagged at the Willow Creek seining station on September 28 and then recaptured two days later at Sheephouse Creek station. The second fish was tagged at Jenner Gulch on June 22 and recaptured at the same site three days later. No steelhead tagged upstream of the Estuary were later recaptured in the Estuary. The size and growth patterns of steelhead are shown in Figure 4.4.9.

Chinook Salmon

A total of 194 Chinook salmon smolts were captured by beach seine in the Estuary during 2015 (Table 4.4.1). The abundance of smolts in the Estuary has varied since studies began in 2004 (Figure 4.4.10). Chinook salmon abundance was lowest in 2005, 2012, and 2013 at 0.7 fish/set.

The highest abundance of Chinook salmon smolts was in 2008 at 4.6 fish/set. The CPUE in 2015 was moderately low at 1.3 fish/set. Chinook salmon smolts are usually most abundant during May and June (Figure 4.4.11) and rarely encountered after July. Monthly smolt captures in 2015 were highest during May at 3.8 fish/set. Chinook salmon smolts were distributed throughout the Estuary with captures at most sample stations and reaches annually (Figure 4.4.12).

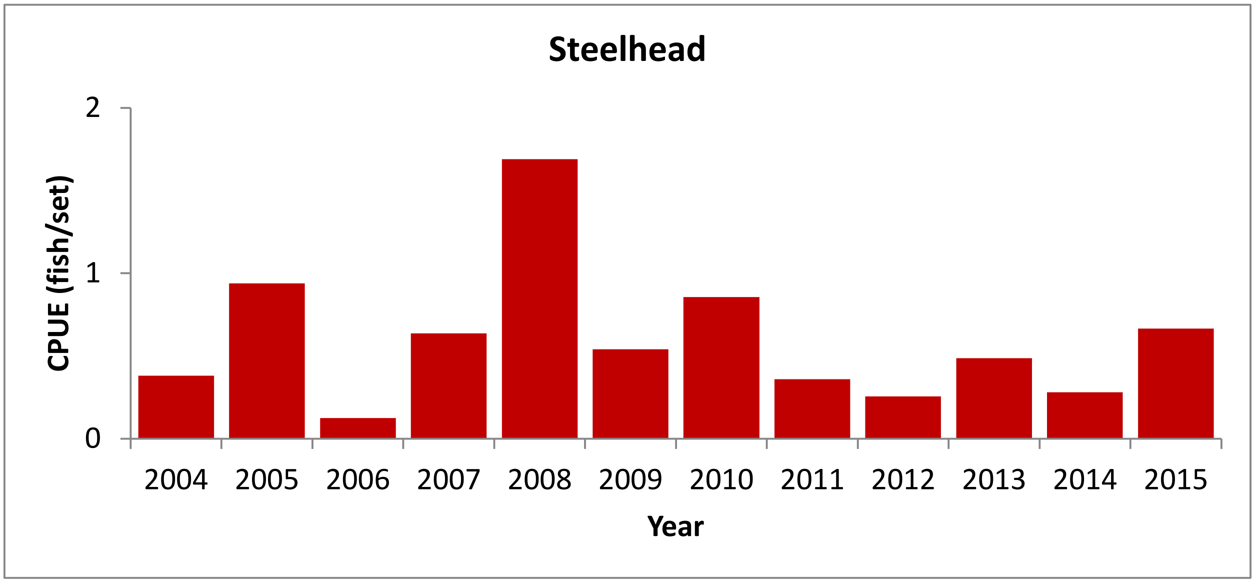


Figure 4.4.4. Annual abundance of juvenile steelhead captured by beach seine in the Russian River Estuary, 2004-2015. Samples are from 96 to 300 seine sets conducted yearly between May and October.

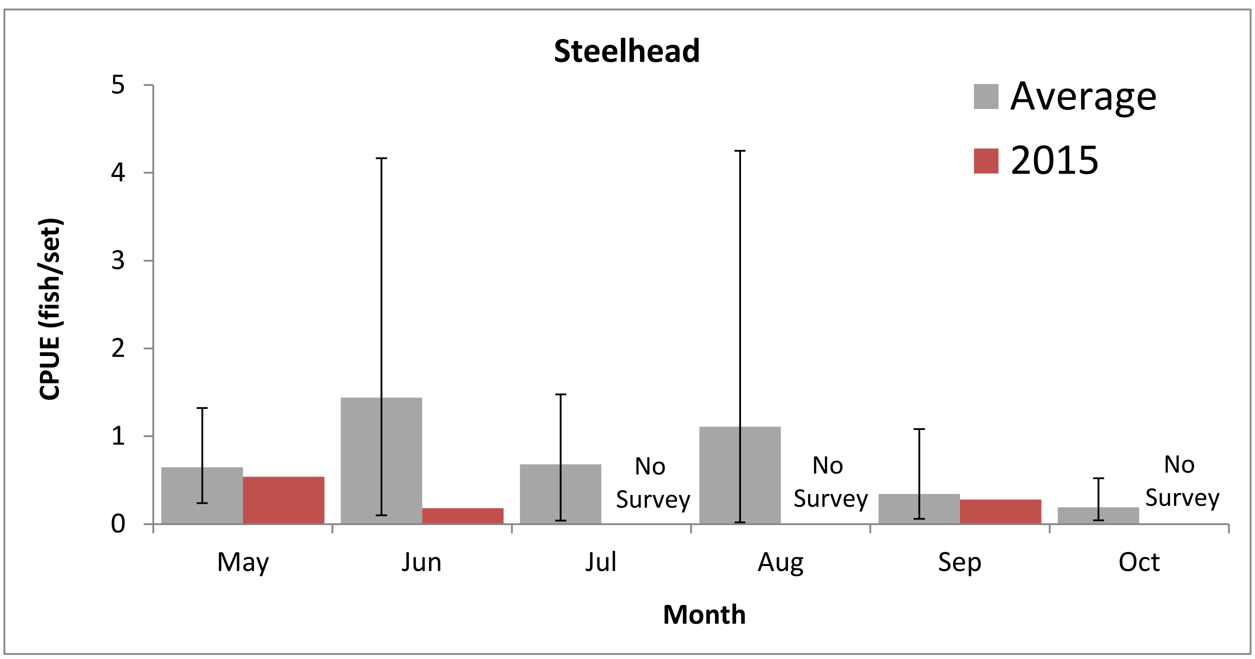


Figure 4.4.5. Seasonal abundance of juvenile steelhead captured by beach seine in the Russian River Estuary, 2004-2015. Seining events consisted of 21 to 50 seine sets approximately monthly. October surveys began in 2010. Data from 2004 to 2014 were averaged and whiskers indicate minimum and maximum values.

Figure 4.4.6. Distribution of juvenile steelhead captured by beach seine in the Russian River Estuary, 2004-2015. Fish were sampled by beach seine consisting of 96 to 300 sets annually. No surveys were conducted in the Upper2 Sub-Reach (Casini Ranch and Moscow Bridge stations) from 2004 to 2009. Data from 2004 to 2014 were averaged and whiskers indicate minimum and maximum values.

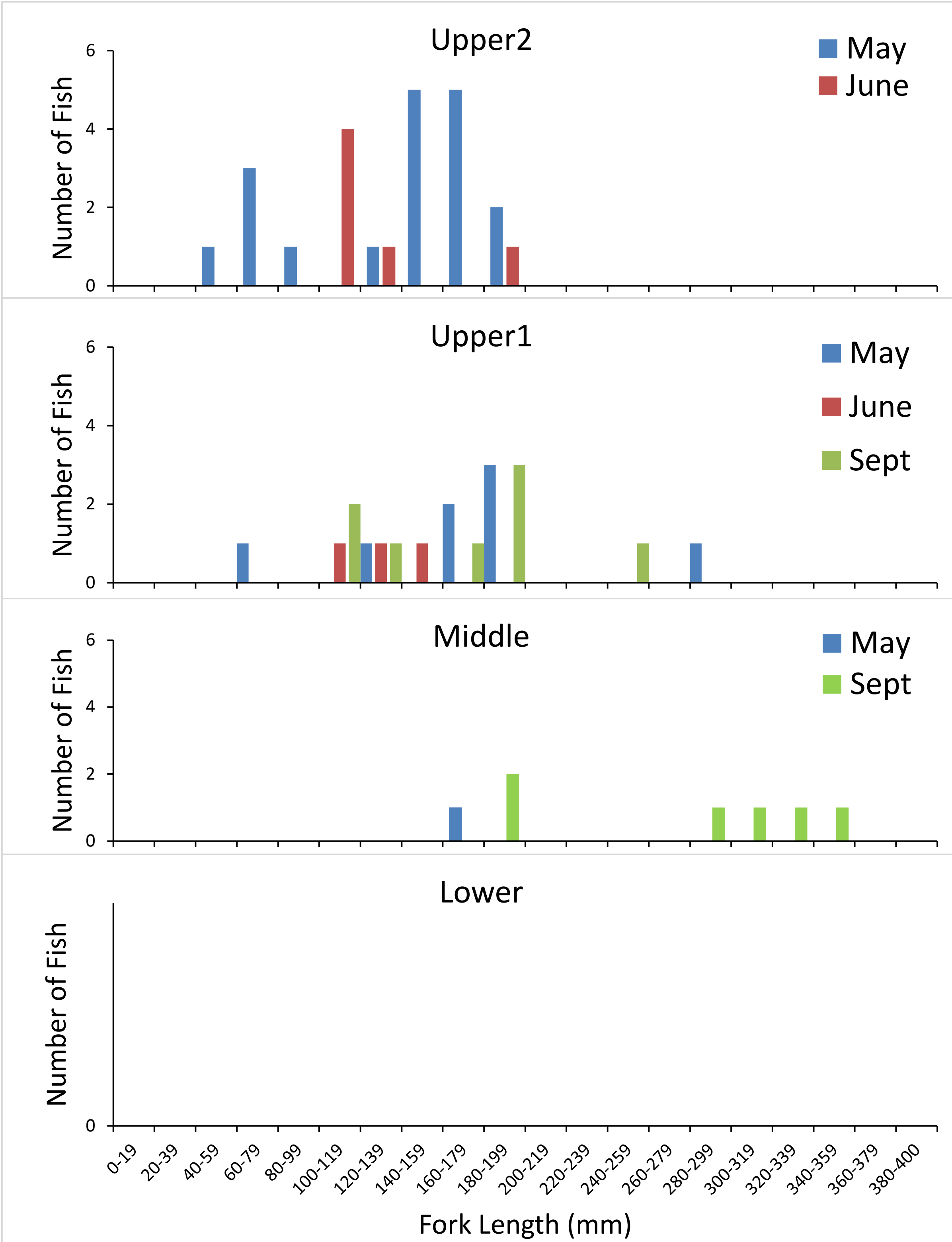


Figure 4.4.7. Length frequency of juvenile steelhead captured by beach seine in the Russian River Estuary, 2015. Fish captures are grouped by Estuary reach and month.

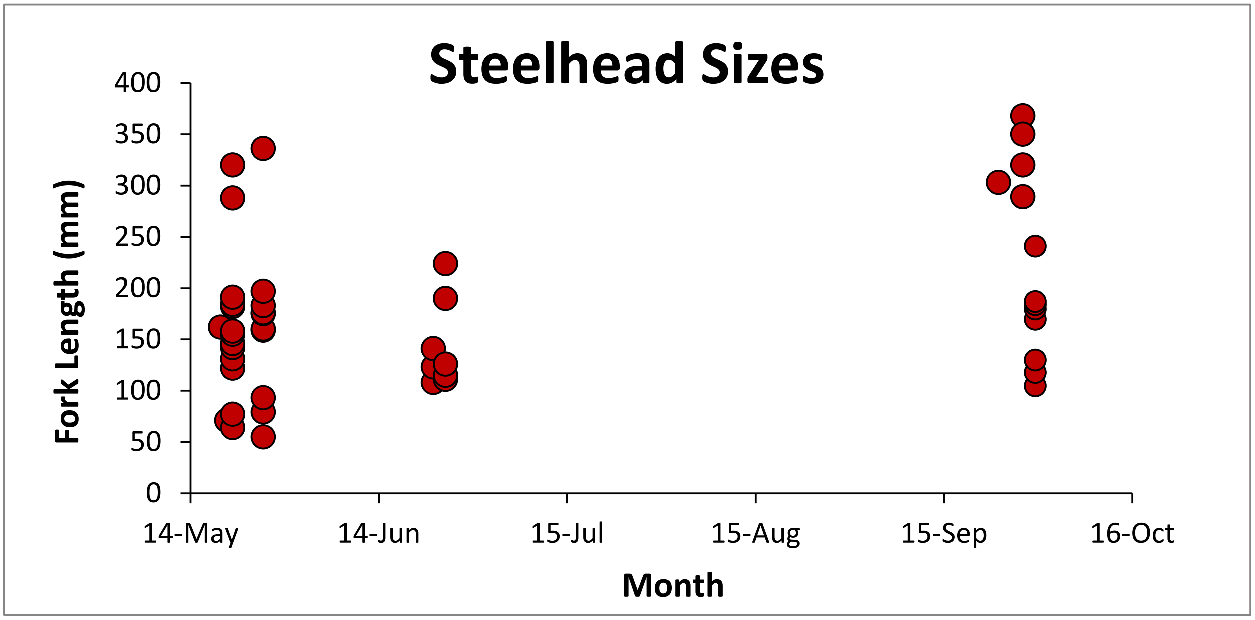


Figure 4.4.8. Juvenile steelhead sizes captured by beach seine in the Russian River Estuary, 2015.

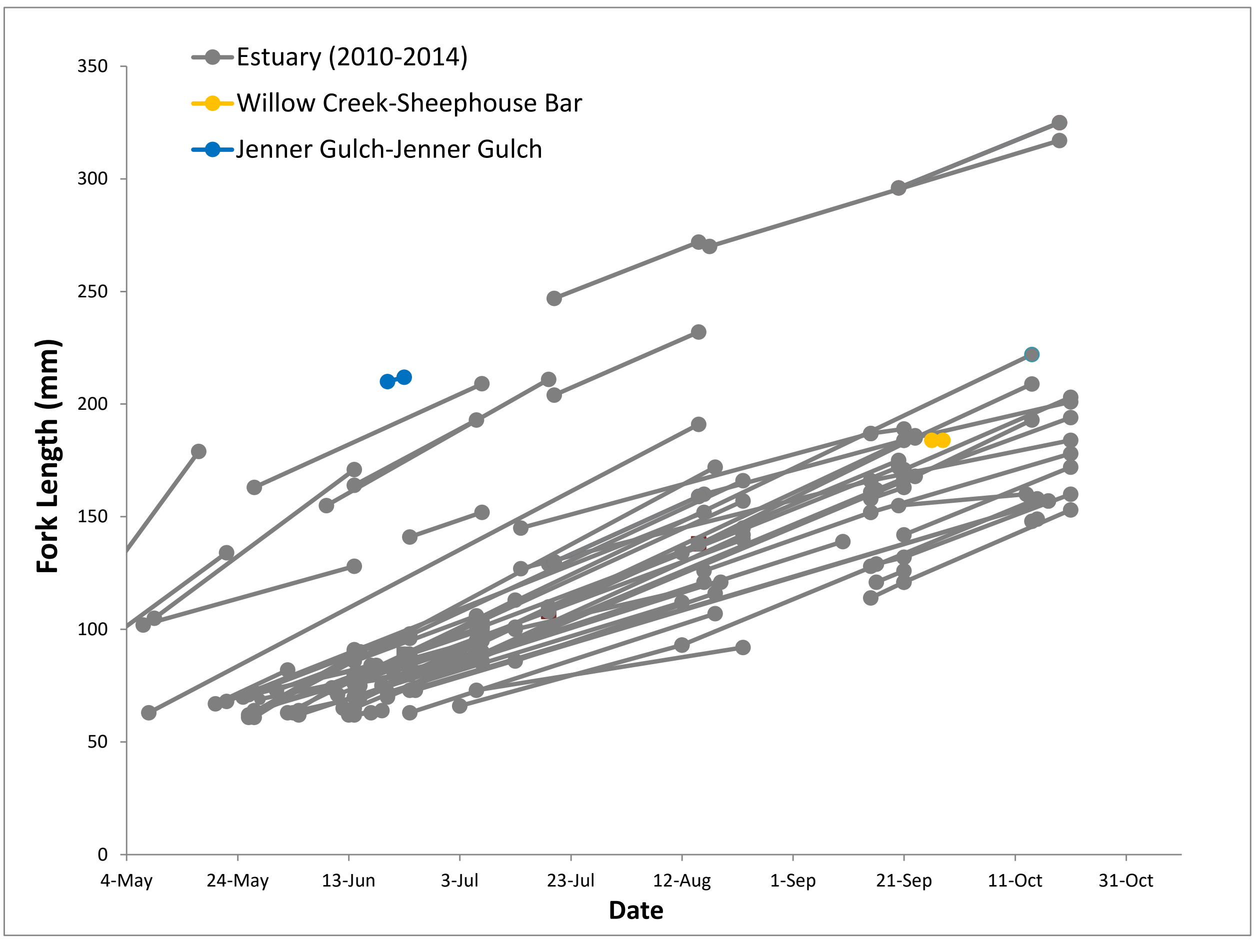


Figure 4.4.9. Growth rates of juvenile steelhead in the Estuary, 2010-2015. Fish were either PIT tagged in the Estuary or upstream and then recaptured in the Estuary. Fish from 2010-2014 are shown in gray. All other colors are steelhead from 2015.

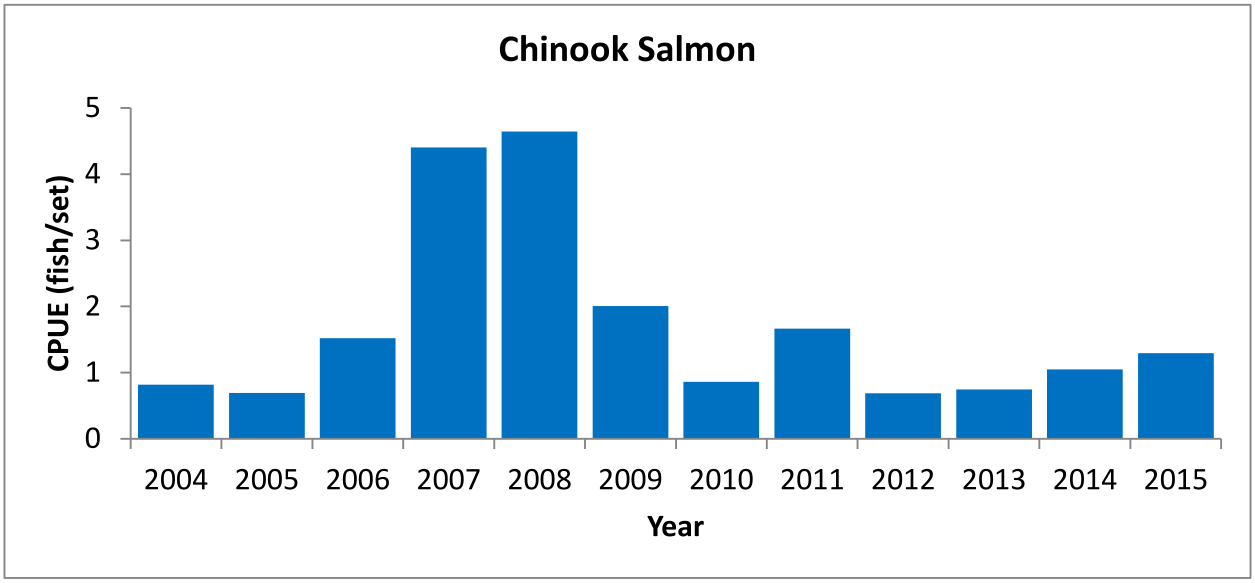


Figure 4.4.10. Annual abundance of Chinook salmon smolts captured by beach seine in the Russian River Estuary, 2004-2015. Samples are from 96 to 300 seine sets yearly between May and October.

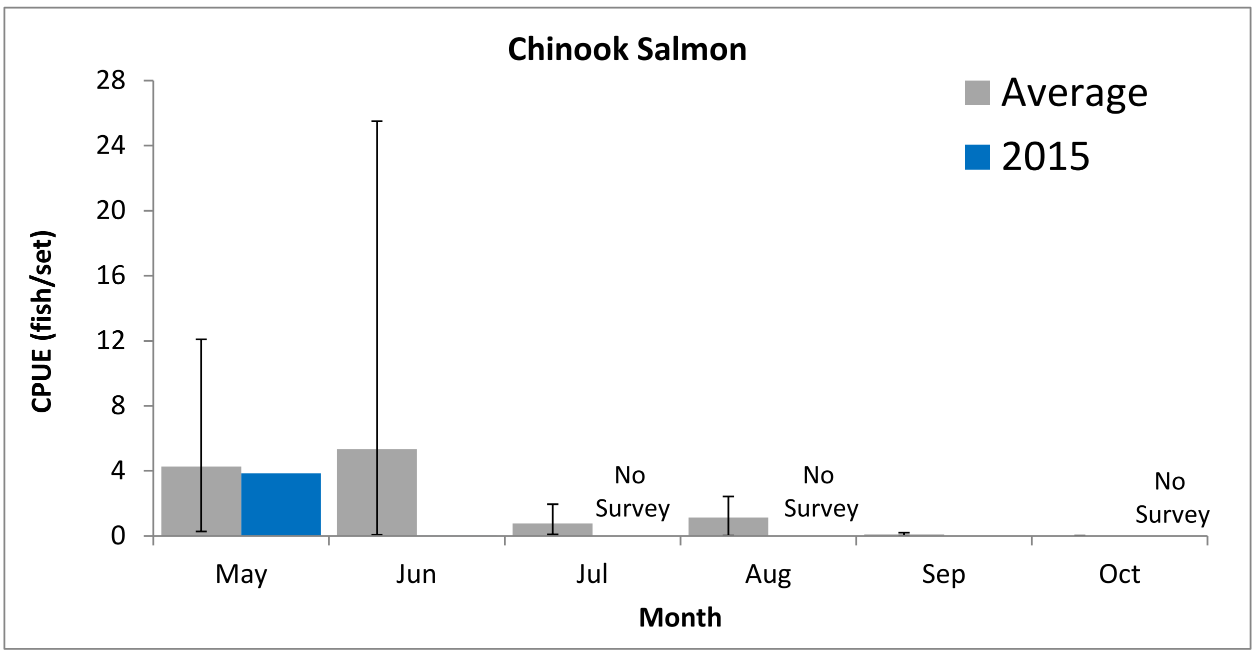


Figure 4.4.11. Seasonal abundance of Chinook salmon smolts captured by beach seine in the Russian River Estuary, 2004-2015. Seining events consisted of 21 to 50 seine sets approximately monthly. October surveys began in 2010. Data from 2004 to 2014 were averaged. Whiskers indicate minimum and maximum values.

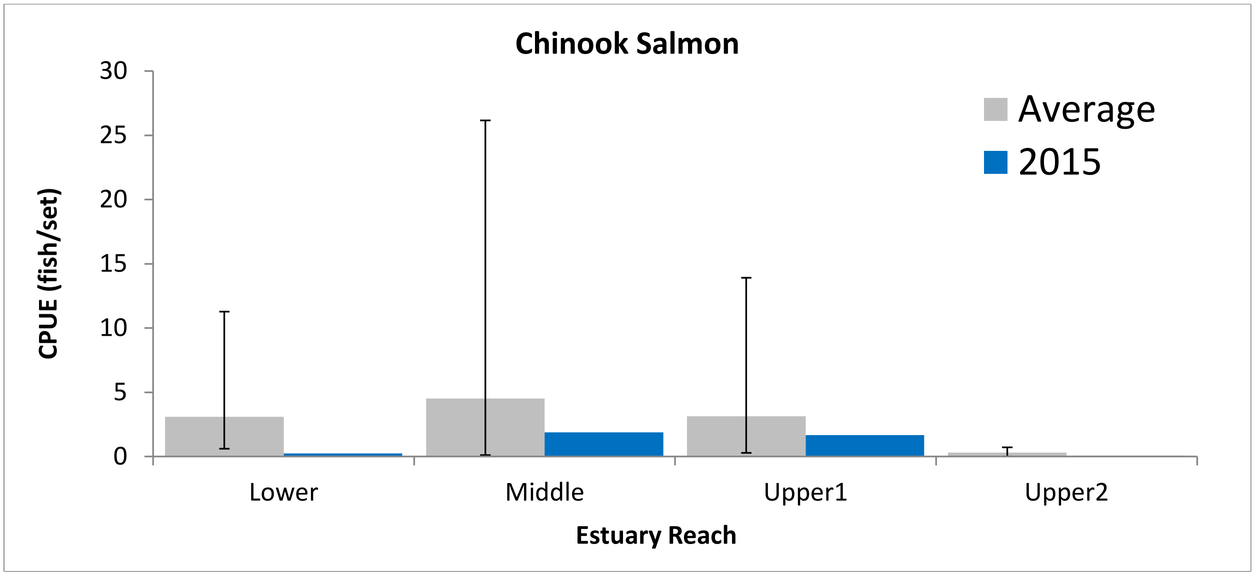


Figure 4.4.12. Spatial distribution of Chinook salmon smolts in the Russian River Estuary, 2004-2015. Fish were sampled by beach seine consisting of 96 to 300 sets annually. Data from 2004 to 2014 were averaged. Whiskers indicate minimum and maximum values. No surveys were conducted in the Upper2 Sub-Reach (Casini Ranch and Moscow Bridge stations) from 2004 to 2009.

There were three Chinook smolts PIT-tagged in Dry Creek at a downstream migrant trap station that were recaptured in the Middle Estuary Reach. Transit times were 6 to 7 days to move downstream 55 rkm. These smolts ranged in size from 90 to 96 mm fork length when captured in the Estuary and had a growth rate of 0.3 to 0.9 mm/d.

Coho Salmon

There have been relatively few coho salmon smolts captured in the Estuary during our beach seining surveys (Figure 4.4.13). The first coho salmon smolt captured in the Estuary was a single fish in 2006. In 2011 and 2015 there were marked increases in abundances of coho smolts with a CPUE of 0.9 and 0.7 fish/set, respectively. During 2015 the total capture of coho smolts was 106, which is the second largest of all study years. The relatively low coho salmon captures in the Estuary are related to their scarcity in the Russian River watershed, but also the timing of our seining surveys that begin in late-May or June when most smolts have already migrated to the ocean. Nearly all coho salmon smolts were captured by June (Figure 4.4.14). The spatial distribution of coho smolts has varied annually (Figure 4.4.15). In 2015 coho were captured in all reaches, except Upper2 Sub-Reach.

Twelve of the Estuary-captured coho salmon were PIT-tagged hatchery fish (Mariska Obedzinski, UC extension, unpublished data). These hatchery coho were stocked in several tributaries of the Russian River including Dry Creek, Pena Creek (tributary to Dry Creek), and Green Valley Creek. Also, coho were stocked in Willow Creek, a tributary to the Middle Reach of the Estuary. Release dates ranged from winter of 2014/2015 to spring 2015. Also, two coho where released during summer and fall 2014. Time from release in creeks to capture in the

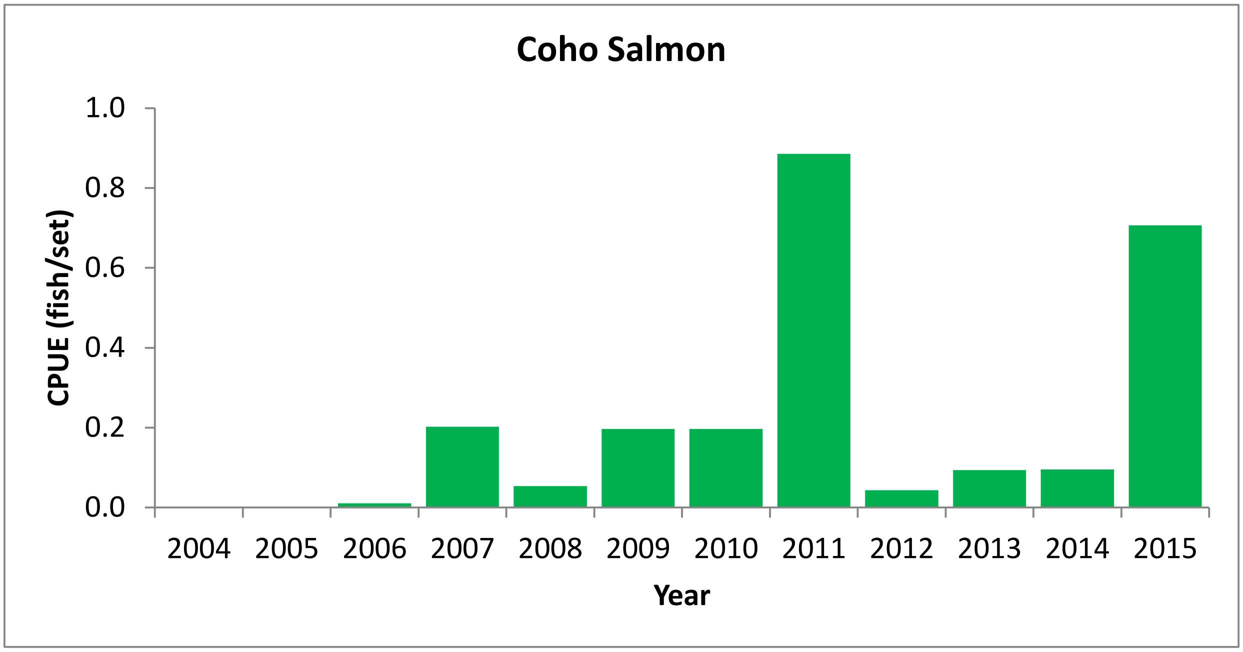


Figure 4.4.13. Annual abundance of coho salmon smolts captured by beach seine in the Russian River Estuary, 2004-2015. Samples are from 96 to 300 seine sets yearly from May to October.

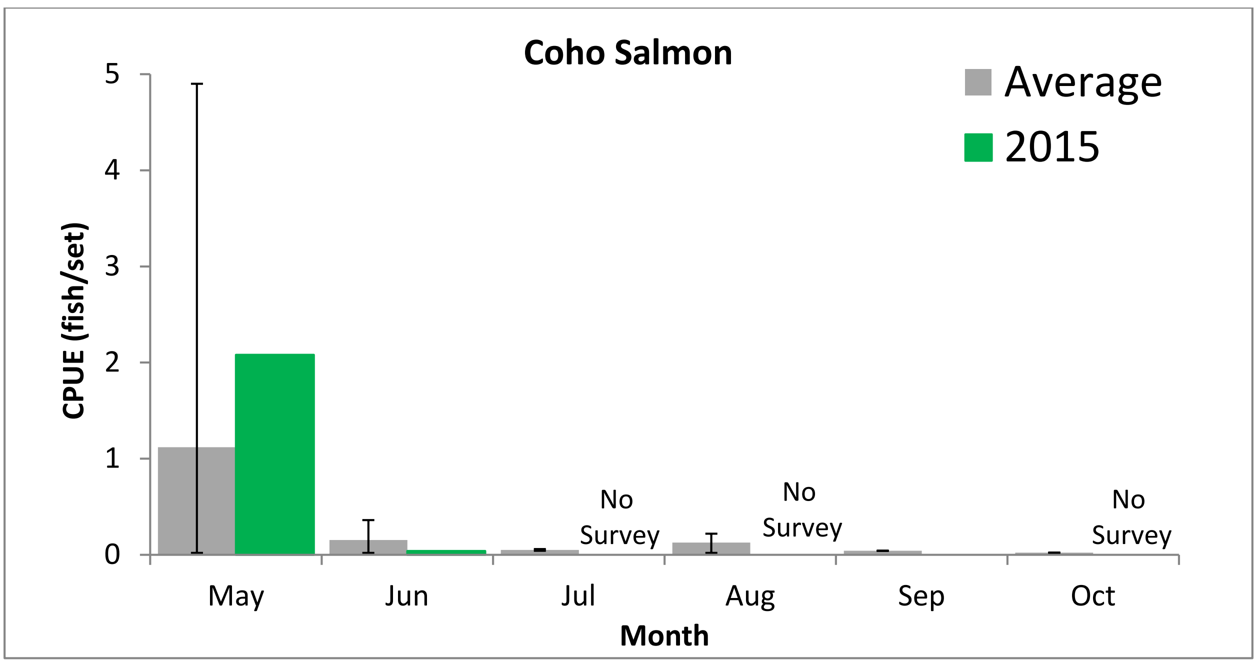


Figure 4.4.14. Seasonal abundance of coho salmon smolts captured by beach seine in the Russian River Estuary, 2004-2015. Seining events consisted of 21 to 50 seine sets approximately monthly. October surveys began in 2010. Data from 2004 to 2014 were averaged. Whiskers indicate minimum and maximum values.

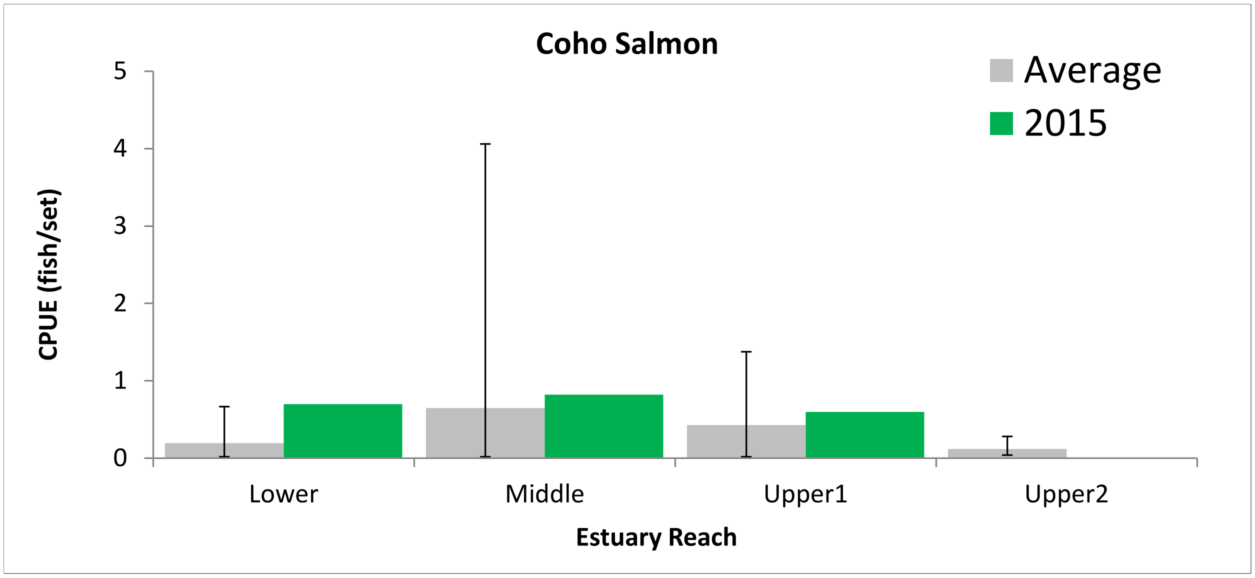


Figure 4.4.15. Spatial distribution of coho salmon smolts in the Russian River Estuary, 2004-2015. Fish were sampled by beach seine consisting of 96 to 300 sets annually. No surveys were conducted in the Upper2 Sub-Reach (Casini Ranch and Moscow Bridge stations) from 2004 to 2009. Data from 2004 to 2014 were averaged. Whiskers indicate minimum and maximum values.

Estuary ranged from 12 to 351 days. The variation in movement patterns of coho is described by the following three tagged fish:

- Green Valley Creek: hatchery coho PIT# 3DD.003BCDC3C0 was released in Green Valley Creek on February 17, 2015, then was recorded at the Green Valley antenna station, located downstream, several times between April 20 and 27 before departing the creek on April 28. This fish was then captured by seine at the Russian River mouth on May 13. After release and spending two months in Green Valley Creek the transit time to the river mouth at the Pacific Ocean took 15 days covering 40 rkm.
- Pena Creek: hatchery coho PIT # 3DD.003BCE01EA was released in Pena Creek (tributary to Grape and Dry creeks) on December 10, 2014. It was detected at three antenna stations in the Dry Creek watershed between April 27 and 28, in a downstream direction, and arrived in the Upper Estuary (Duncans Mills antenna) on April 30. Then the smolt was captured by seine at the river mouth on May 13. This smolt traveled from the headwaters of Dry Creek watershed to the upper Estuary in three days covering 54 rkm and then spent 13 days to move 10 rkm to the river mouth.
- Willow Creek: hatchery coho PIT # 3DD.003BCE1653 was released in Willow Creek (tributary to the Middle Estuary) on June 11, 2014. It was detected at an antenna station downstream of its release site on May 2-3, 2015 and then passed an antenna station near the confluence with the Middle Estuary on May 12-13. The smolt was captured at the Bridgehaven seining station, downstream of Willow Creek, on May 19, 2015. This coho moved 4 rkm in 16 days from Willow Creek to the Middle Estuary.

The growth rates of coho smolt increased from winter to spring. Growth rates of four hatchery coho ranged from 0.1 to 0.4 mm/d that were released in Russian River tributaries in February 2015 and captured in the Estuary in May and June. In comparison, four coho released or captured in Russian River tributaries in April 2015 and then (re)captured in the Estuary one month later in May had faster growth at 0.3 to 0.8 mm/d. It is unclear if the acceleration of coho growth in the spring occurred while in the Russian River and its tributaries and/or in the Estuary.

American Shad

American shad is an anadromous sportfish, native to the Atlantic coast. It was introduced to the Sacramento River in 1871 and within two decades was abundant locally and had established populations from Alaska to Mexico (Moyle 2002). Adults spend from 3 to 5 years in the ocean before migrating upstream to spawn in the main channels of rivers. Juveniles spend the first year or two rearing in rivers or estuaries.

The abundance of American shad in the Estuary during 2015 was low at 1.0 fish/set (Figure 4.4.16). This low abundance may have been influenced by the reduced seining effort in 2015 where no surveys were conducted during July and August. Typically, juvenile American shad first appear in relatively large numbers in July and the catch usually peaks in August. Shad are typically distributed throughout the Estuary, although in 2015 they were only found in the Upper1 Sub-Reach (Figure 4.4.17).

Topsmelt

Topsmelt are one of the most abundant fish in California estuaries (Baxter et al. 1999) and can tolerate a broad range of salinities and temperatures, but are seldom found in freshwater (Moyle 2002). They form schools and are often found near the water surface in shallow water. Sexual maturity is reached in 1 to 3 years and individuals can live as long as 7 to 8 years. Estuaries are used as nursery and spawning grounds and adults spawn in late spring to summer.

Topsmelt is a common fish in the Russian River Estuary. However, the abundance of topsmelt in the Estuary has varied substantially since 2004. After a peak in 2006 with a CPUE of 13.4 fish/set the abundance of topsmelt decreased until 2012 with a CPUE of 0.3 fish/set (Figure 4.4.18). Since 2012 the abundance of topsmelt has been high, including the highest CPUE recorded at 22.2 fish/set in 2014. The abundance in 2015 was moderately high at 7.0 fish/set. Also, the abundance of topsmelt in 2015 may been an underestimate because no seining was conducted in July and August when the catch of topsmelt usually peaks. Topsmelt are mainly distributed in the Lower and Middle Reaches in the Estuary (Figure 4.4.19).

Starry Flounder

Starry flounder range from Japan and Alaska to Santa Barbara in coastal marine and estuarine environments. In California, they are common in bays and estuaries (Moyle 2002). This flatfish is usually found dwelling on muddy or sandy bottoms. Males mature during their second year and females mature at age 3 or 4 (Baxter et al. 1999). Spawning occurs during winter along the coast, often near the mouths of estuaries. Young flounders spend at least their first year rearing in estuaries. They move into estuaries during the spring and generally prefer warm, low-salinity water or freshwater. As young grow, they shift to using brackish waters.

The abundance of juvenile starry flounder in the Estuary has generally decreased since 2004 and 2005 (Figure 4.4.20). Juvenile flounder have been at relatively low abundance since 2006. The CPUE in 2015 was 1.5 fish/set. The Estuary appears to be utilized primarily by young-of-

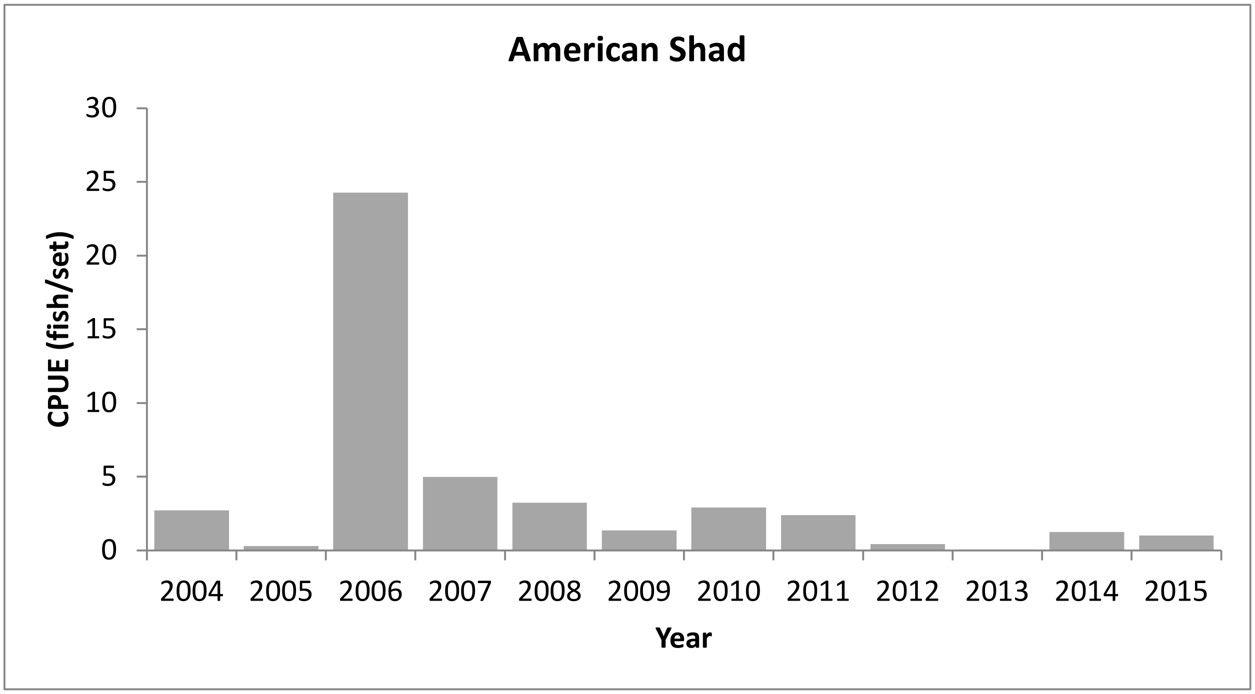


Figure 4.4.16. Annual abundance of juvenile American shad captured by beach seine in the Russian River Estuary, 2004-2015. Samples are from 96 to 300 seine sets yearly from May to October.

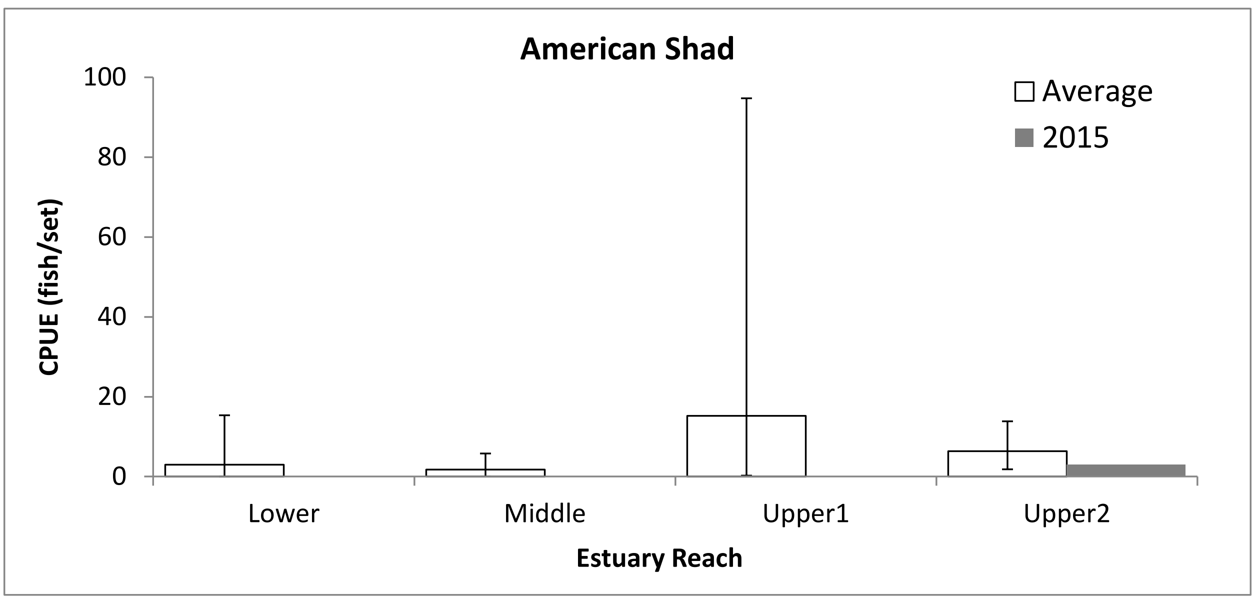


Figure 4.4.17. Spatial distribution of juvenile American shad in the Russian River Estuary, 2004-2015. Fish were sampled by beach seine consisting of 96 to 300 sets annually. No surveys were conducted in the Upper2 Reach during 2004 and 2009. Data from 2004 to 2014 were averaged. Whiskers indicate minimum and maximum values.

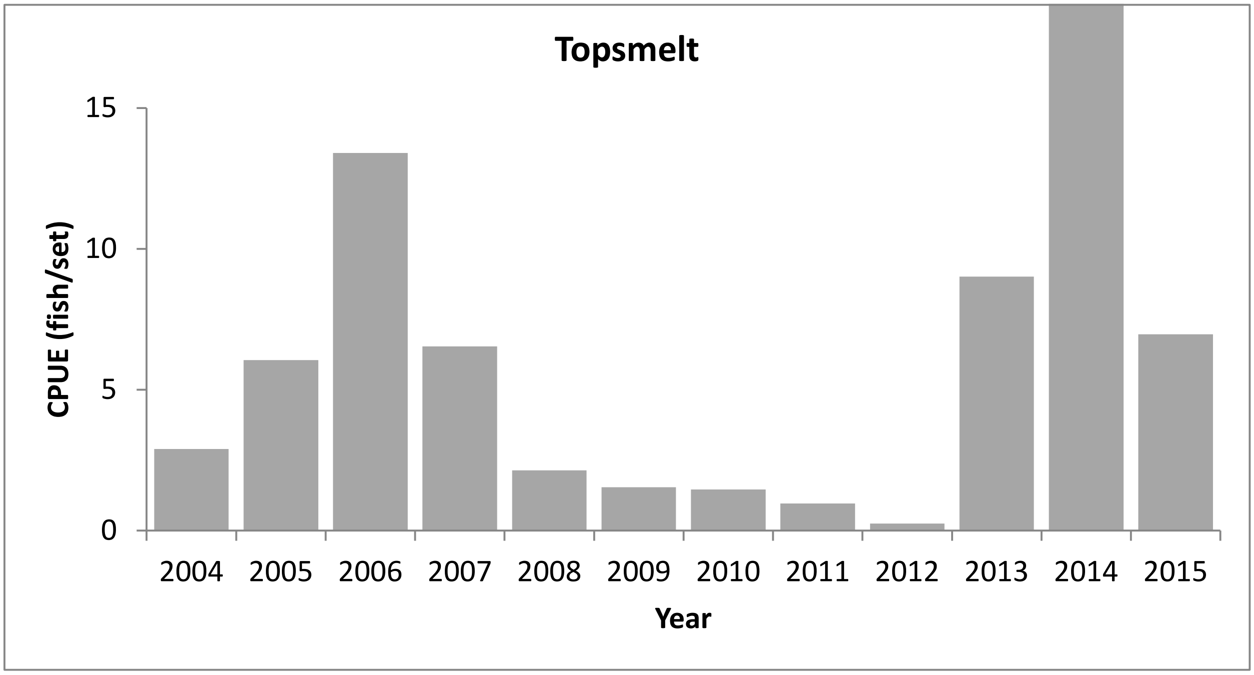


Figure 4.4.18. **Annual abundance of topsmelt captured by beach seine in the Russian River Estuary, 2004- 2015. Samples are from 96 to 300 seine sets yearly from May to October.**

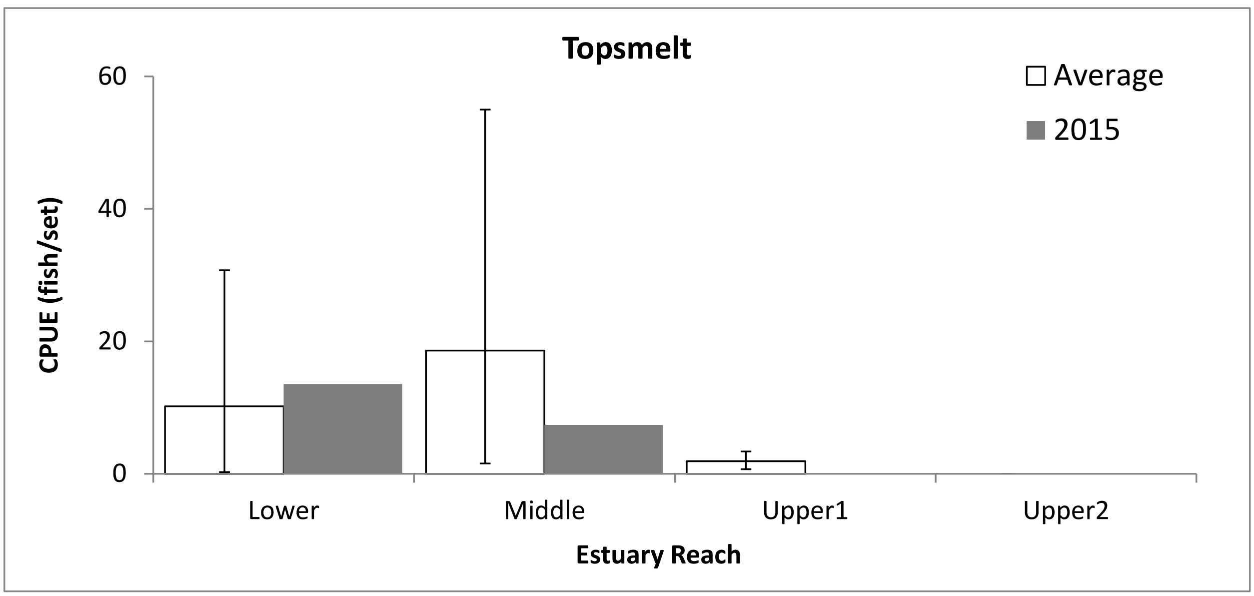


Figure 4.4.19. Spatial distribution of topsmelt in the Russian River Estuary, 2004-2015. Fish were sampled by beach seine consisting of 96 to 300 sets annually. No surveys were conducted in the Upper2 Reach during 2004 and 2009. Data from 2004 to 2014 were averaged. Whiskers indicate minimum and maximum values.

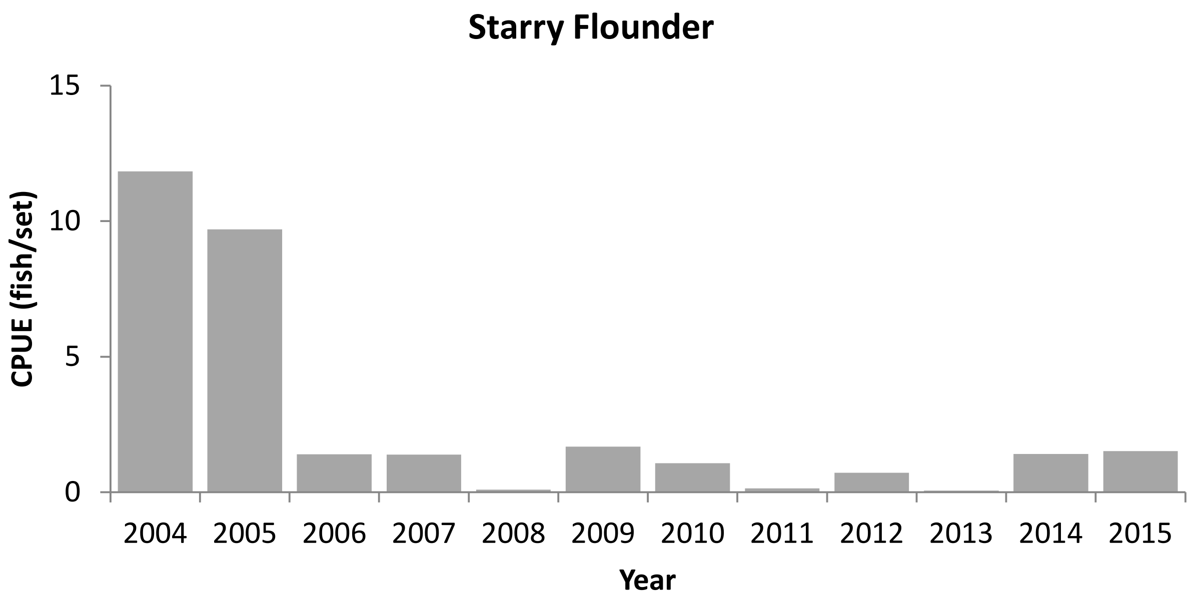


Figure 4.4.20. Annual abundance of juvenile starry flounder captured by beach seine in the Russian River Estuary, 2004-2015. Samples are from 96 to 300 seine sets yearly from May to October.

the-year fish where most flounder captures are less than 100 mm fork length. The seasonal occurrence of starry flounder was typically highest in May and June, and then gradually decreased through September and October when few were caught. Starry flounder were distributed throughout the Estuary ranging from the River Mouth in the Lower Reach, with cool seawater conditions, to the Upper Reach, with warm freshwater (Figure 4.4.21). Starry flounder have been detected as far as Austin Creek at the upstream end of the Estuary (Cook 2006).

Conclusions and Recommendations

Fish Sampling - Beach Seining

The results of Estuary fish surveys from 2004 to 2015 found a total of 50 fish species from marine, estuarine, and riverine origins. The distribution of species was strongly influenced by the salinity gradient in the Estuary that is typically cool seawater near the mouth of the Russian River and transitions to warmer freshwater at the upstream end. Exceptions to this distribution pattern were anadromous and generalist fish that occurred throughout the Estuary regardless of salinity levels. A late season river mouth closure occurred in September/October 2015 that formed a lagoon. In response to changing water conditions the distribution of fish shifted. Estuarine species were more concentrated in the Lower Reach with higher salinity and freshwater species moved upstream into the Upper2 Reach, which was primarily fresh. This

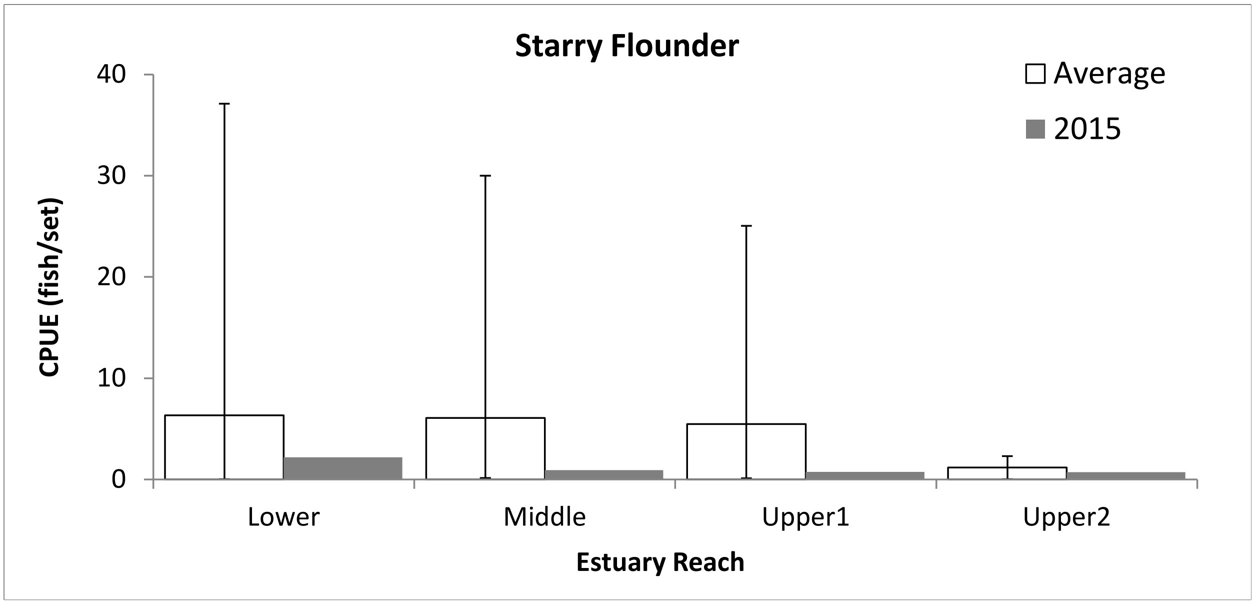


Figure 4.4.21. Spatial distribution of juvenile starry flounder in the Russian River Estuary, 2004-2015. Fish were sampled by beach seine consisting of 96 to 300 sets annually. No surveys were conducted in the upper Estuary during 2004 and 2009. Data from 2004 to 2014 were averaged. Whiskers indicate minimum and maximum values.

short duration lagoon forming late in the season appeared to have no effect on the abundance of steelhead in the Estuary (Figure 4.4.7).

The results of the 2015 fish studies contribute to the 12-year dataset of existing conditions and our knowledge of a tidal brackish system. This baseline data will be used to compare with a closed mouth lagoon system. However, until a prolonged lagoon is formed reducing the seining effort may be acceptable as was the case in 2013 to 2015 when seining surveys were conducted in May, June, and September.

The distribution and abundance of salmonids in the Estuary differed spatially, temporally, and by species. Steelhead were usually captured from May to October during each study year. PIT-tagged steelhead showed strong fidelity to specific sites in the Estuary and grew rapidly. This indicates that steelhead rear in the Estuary under current river mouth management conditions.

The fluctuation in abundance of steelhead annually is likely attributed to the variability in adult spawner population size (i.e. cohort abundance), residence time of young steelhead before out-migration, and schooling behavior that affects susceptibility to capture by seining. Chinook salmon smolts spent less than half the summer rearing in the Estuary and were usually absent after July. Based on the detection of these smolts at most seining stations, they appear to use most estuarine habitats as they migrate to the ocean. In comparison, steelhead were found during the entire summer and were often found in the Upper Reach of the Estuary. However, there are sites in the Middle and Lower Estuary (e.g., Jenner Gulch confluence) where steelhead are consistently found.

Although beach seining is widely used in estuarine fish studies, beach seines are only effective near shore in relatively open water habitats free of large debris and obstructions that can foul or snag the net. Consequently, there is inherent bias in seine surveys (Steele et al. 2006). By design, our seining stations were located in areas with few underwater obstructions (i.e., large rocks, woody debris, etc.) and this likely influenced our assessment of fish abundance and habitat use. However, the spatial and temporal aspects of our sampling do allow quantitative comparisons among reaches and years.

4.5 Downstream Migrant Trapping

The Reasonable and Prudent Alternative (RPA) in the Russian River Biological Opinion compels the Water Agency to provide information about the timing of downstream movements of juvenile steelhead, their relative abundance and the size/age structure of the population as related to the implementation of an adaptive management approach to promote formation of a perched freshwater lagoon. The sampling design implemented by the Water Agency and described in this section specifically targets the detection and capture of anadromous salmonid young-of-the-year (YOY, age-0) and parr (≥age-1) (collectively referred to as juveniles) as well as smolts. In order to help accomplish the objectives listed above, the Water Agency undertook fish capture and PIT-tagging activities at selected trapping sites upstream of the estuary (Figure 4.5.1):

- Dry Creek (capture only)
- Mainstem Russian River at Mirabel (not operated in 2015)
- Mark West Creek
- Dutch Bill Creek
- Austin Creek

Stationary PIT antenna arrays were operated in the following locations:

- Mainstem Russian River at Northwood (19.16 rkm)
- Upstream end of the Russian River Estuary in Duncans Mills (10.46 rkm)
- Near the mouth of Austin Creek (0.5 rkm)

Implementation of the monitoring activities described here are the result of a continually-evolving process of evaluating and improving on past monitoring approaches. Descriptions and data from other monitoring activities conducted in the estuary (e.g., water quality monitoring, beach seining) as well as fish trapping operations in Dry Creek are presented elsewhere.

Methods

In 2015 we again relied on downstream migrant traps (DSMT) and stationary PIT antenna arrays at lower-basin trap sites to address the objectives in the RPA. Similar to 2010 through 2014, fish were physically captured at downstream migrant traps (rotary screw trap, funnel trap or pipe trap depending on the site), sampled for biological data and released. PIT tags were applied to a subset of age-0 steelhead captured at trap sites and fish were subject to detection at downstream PIT antenna arrays if they moved downstream into the estuary. In the sections that follow, we describe the sampling methods and analyses conducted for data collected at each site.

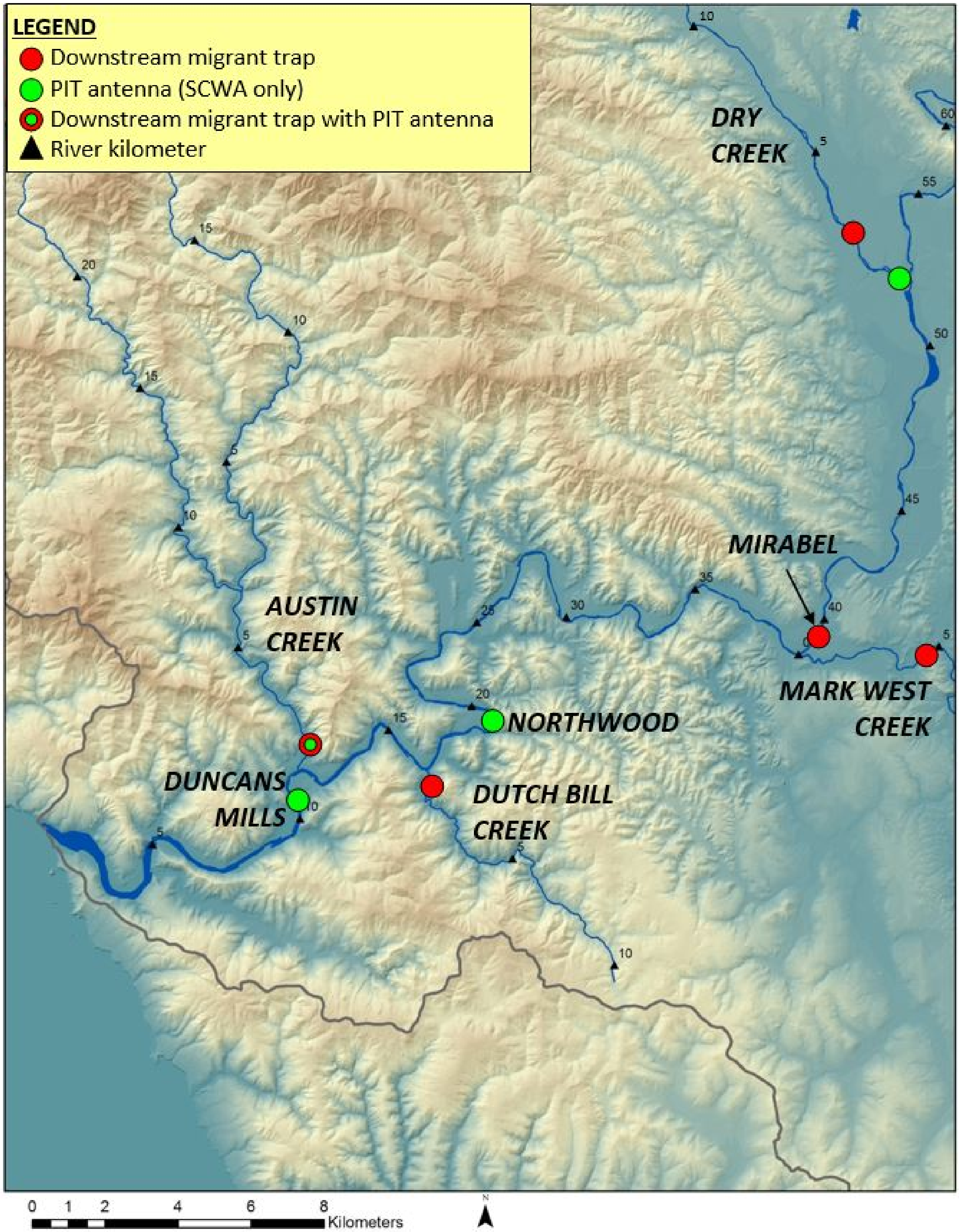


Figure 4.5.1. Downstream migrant detection sites in the lower Russian River, 2015. Numbered symbols along stream courses represent distance (rkm) from the mouth of each stream.

Estuary/Lagoon PIT antenna systems

Two antenna arrays with multiple flat plate antennas (antennas designed to lay flat on the stream bottom) were installed in the upper Russian River Estuary (Estuary) near the town of Duncans Mills (rkm 10.44 and 10.46) to detect PIT-tagged fish entering the estuary (Figure 4.5.2). Generally, 12 antennas were operated continuously from January 1 until May 28 (the period during which Austin Creek remained connected to the mainstem Russian River by surface flow). The orientation of the antennas consisted of 2 rows of six antennas with one row slightly upstream of the other. Each row contained 6 antennas placed side by starting at the west river bank and extending out into the channel.

Figure 4.5.2. Flat plate antenna arrays at Duncans Mills (rkm 10.44 and 10.46). Rectangles represent individual flat plate antennas.

In 2013 and 2014, a dual flat plate PIT antenna array was operated in the mainstem Russian River in the vicinity of the golf course near the community of Northwood. The objective of this effort was to provide a means of detecting movements of juvenile steelhead that were PIT-tagged at upstream trap sites that may move into that portion of the mainstem of the Russian River that is non-tidal but can be inundated under perched lagoon or closed river mouth conditions. The antenna array consisted of two PIT antennas oriented so that they spanned approximately 75% of the wetted width of the river including the entire thalweg during open-mouth/non-perched conditions.

Lower River Fish Trapping and PIT tagging

Following consultation with NMFS and CDFW, the Water Agency identified three lower River tributaries (Mark West Creek, Dutch Bill Creek and Austin Creek, Figure 4.5.1) in which to operate fish traps as a way to supplement data collected from the Duncans Mills PIT antenna array and during sampling by beach seining throughout the estuary (Figure 4.5.2). In previous years downstream migrant traps were also operated at the Mirabel inflatable dam. However, a construction project to upgrade the fish ladder and water diversion intake screens precluded us from operating downstream migrant traps at this location in 2015. The Water Agency operated three types of downstream migrant traps depending on the stream, water depth, and velocity: rotary screw trap, funnel trap and pipe trap (Figure 4.5.3). Fish traps were checked daily by Water Agency staff during the trapping season (March through July). Captured fish were

Figure 4.5.3. Photographs of downstream migrant traps operated by the Water Agency. Top: Mark West Creek rotary screw trap (operated March 26-May 18) switched to pipe trap (operated May 19-June 25). Middle: Dutch Bill Creek pipe trap (operated March 19-May 11). Bottom: Austin Creek funnel trap (operated March 25-May 28).

enumerated and identified to species and life stage at all traps. All PIT-tagged fish were measured for fork length (±1 mm) and weighed (±0.1 g). Additionally, a subset of all non-PIT-tagged individuals were measured and weighed each day. PIT tags were implanted in the majority of steelhead YOY and parr captured that were ≥60 mm in fork length.

Mainstem Russian River at Mirabel and Dry Creek at Westside Road

Typically two rotary screw traps (one 5 foot and one 8 foot) adjacent to one another have been operated on the mainstem Russian River immediately downstream of the Water Agency's inflatable dam site at Mirabel (approximately 38.7 rkm upstream of the river mouth in Jenner) (Figure 4.5.1). However, in 2015 active construction of a new fish ladder at Mirabel precluded operating a downstream migrant trap at this location. The Water Agency also operates a 5 foot rotary screw trap in Dry Creek. The purpose of these trapping efforts is to fulfill a broader set of objectives in the Russian River Biological Opinion than what is described in the current section of this report. However, one of the objectives is to provide a source of PIT-tagged steelhead juveniles that may enter the estuary and be detected during downstream monitoring efforts. In 2013 and 2014, 2,702 and 1,353 steelhead YOY and parr were tagged at Dry Creek but very few were detected at downstream locations prior to the end of the lagoon management period on October 15 (seven in both years combined). Based on this experience, steelhead were not tagged at the Dry Creek trap in 2015.

Mark West Creek

A 5-foot rotary screw trap was installed on Mark West Creek approximately 4.8 km upstream of the mouth on March 26. On May 18 the rotary screw trap was removed and replaced with a pipe trap because of low water velocities. The pipe trap was removed and all trapping operations were suspended on June 25 when fish captures dropped off rapidly (Table 4.5.2).

Dutch Bill Creek

A pipe trap was installed on Dutch Bill Creek adjacent to the park in downtown Monte Rio (approximately 0.3 km upstream of the creek mouth) on March 19. The trap was fished until the completion of trapping operations on May 11 when stream flow in lower Dutch Bill Creek became disconnected (Table 4.5.2).

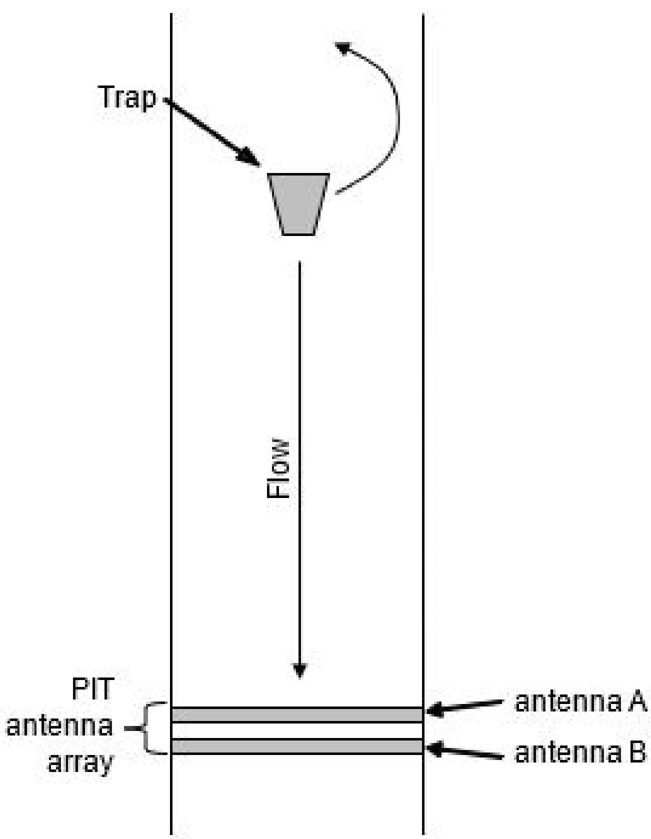
Austin Creek

A funnel trap was installed on Austin Creek on March 25. The funnel trap consisted of wood-frame/plastic-mesh weir panels, a funnel net and a wooden live box. Trapping continued until May 28 when surface flow in lower Austin Creek was no longer contiguous and daily catches of steelhead dropped to zero (Table 4.5.2).

Steelhead parr were marked with PIT tags and released upstream of the trap in order to measure trap efficiency and estimate population size of fish passing the trap site (Figure 4.5.4). We operated a dual PIT antenna array approximately 0.2 km downstream of the funnel trap and approximately 0.5 km upstream from the mouth of Austin Creek in order to detect PIT-tagged steelhead moving out of Austin Creek. The PIT antenna array was located at the upstream extent of the area that can be inundated by the Russian River during closure of the barrier beach; therefore, we assumed that once fish passed the antenna array they had effectively entered the estuary/lagoon. A second PIT tag antenna array located in the Russian River

Table 4.5.2. Installation and removal dates, and total number of days fished for lower river monitoring sites operated by the Water Agency in 2015.

Monitoring site (gear type)	Installation date	Removal date	Number of days fished
Dry Creek (DSMT)	3/18	7/30	121
Mirabel (DSMT)	-	-	0
Mark West Creek (DSMT)	3/26	6/25	586
Northwood (PIT antenna array)	4/28	7/14	77
Dutch Bill Creek (DSMT)	3/19	5/11	49
Austin Creek (DSMT)	3/25	5/28	62
Duncans Mills (PIT antenna array)	continuous (not removed)	continuous (not removed)	entire downstream migration season



1. Methods:
Capture and PIT-tag juvenile steelhead, then release newly tagged fish upstream while releasing previously-tagged fish (recaptures) downstream.

2. Estimating trap efficiency:
Of the PIT-tagged fish released upstream of the trap, how many were recaptured in the trap before being detected on either antenna in the downstream antenna array?

3. Estimating antenna efficiency:
Of the PIT-tagged fish detected on the downstream antenna in the array (antenna B), how many were also detected on the upstream antenna (antenna A).

Figure 4.5.4. Diagram illustrating the relative location of the downstream migrant trap and PIT antenna array operated on Austin Creek and outline of how antenna efficiency was estimated.

estuary at Duncans Mills (approximately 1.5 km downstream) was used to calculate antenna efficiency for the PIT antenna array located in Austin Creek.

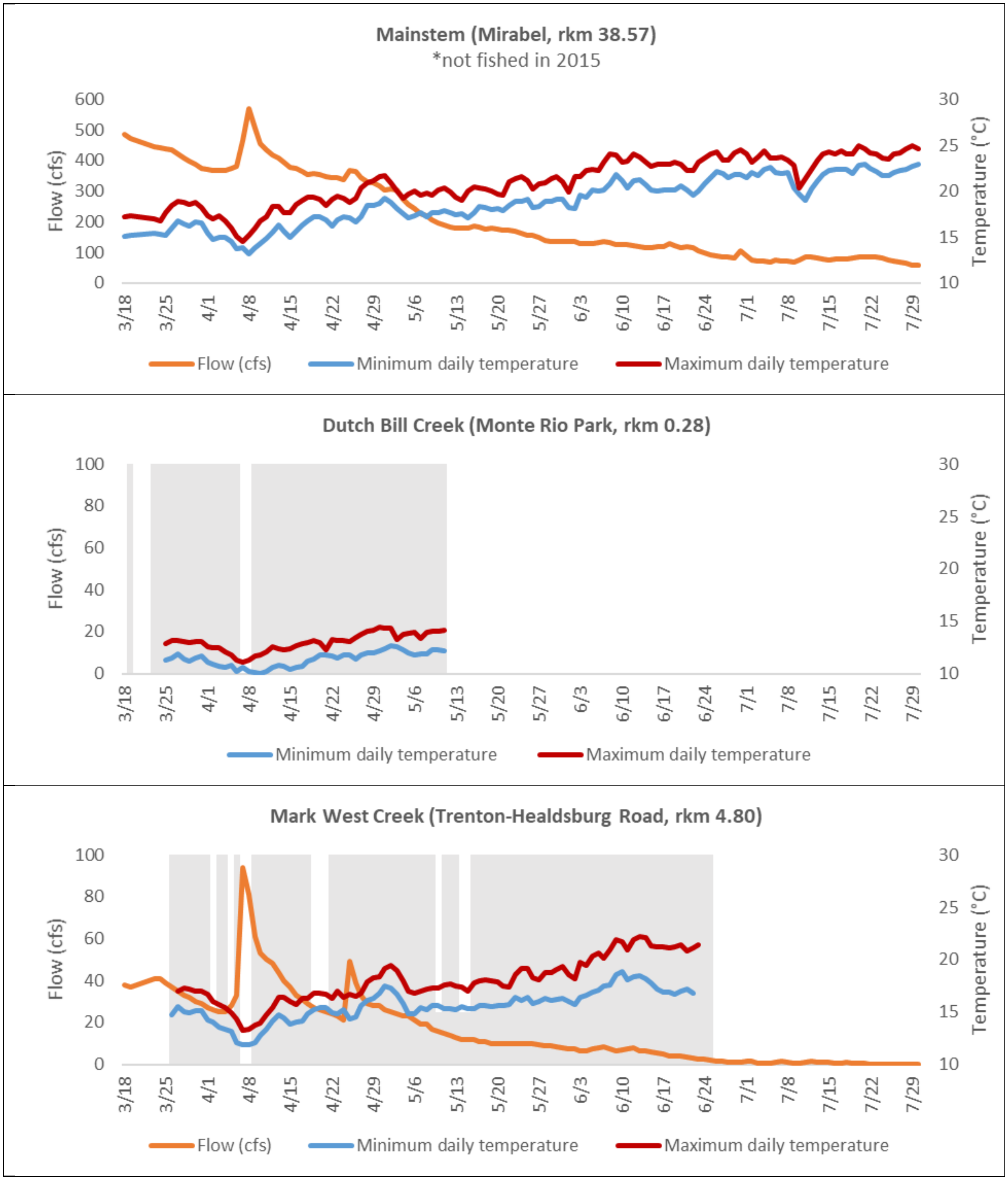
Results

Stream flow largely dictates when downstream migrant traps can be installed (Figure 4.5.5). Our sampling period most likely encompassed a high portion of the juvenile steelhead movement period but we probably missed a substantial portion of the steelhead smolt migration period.

Estuary/Lagoon PIT antenna systems

Steelhead

In Austin Creek 527 juvenile steelhead and 30 smolts were captured while only 16 steelhead were captured in Dutch Bill Creek. At Mark West Creek 47 juveniles and 268 smolts were captured (Figure 4.5.6). The number of fish captured at these traps in 2015 was lower than any other year of trap operation (Table 4.5.3). We believe this was a direct result of low flows which contributed to earlier than low stream connectivity which made downstream movement difficult and could have impacted overall steelhead production from these tributaries. By contrast, at the Dry Creek trap where flows were artificially maintained at a fairly constant level regardless of drought, a total of 4,696 juveniles were captured.


Mainstem (Mirabel, rkm 38.57)
*not fished in 2015
Flow (cfs)
Temperature (°C)
3/18
3/25
4/1
4/8
4/15
4/22
4/29
5/6
5/13
5/20
5/27
6/3
6/10
6/17
6/24
7/1
7/8
7/15
7/22
7/29
Flow (cfs)
Minimum daily temperature
Maximum daily temperature
Dutch Bill Creek (Monte Rio Park, rkm 0.28)
Minimum daily temperature
Maximum daily temperature
Mark West Creek (Trenton-Healdsburg Road, rkm 4.80)
Flow (cfs)
Minimum daily temperature
Maximum daily temperature

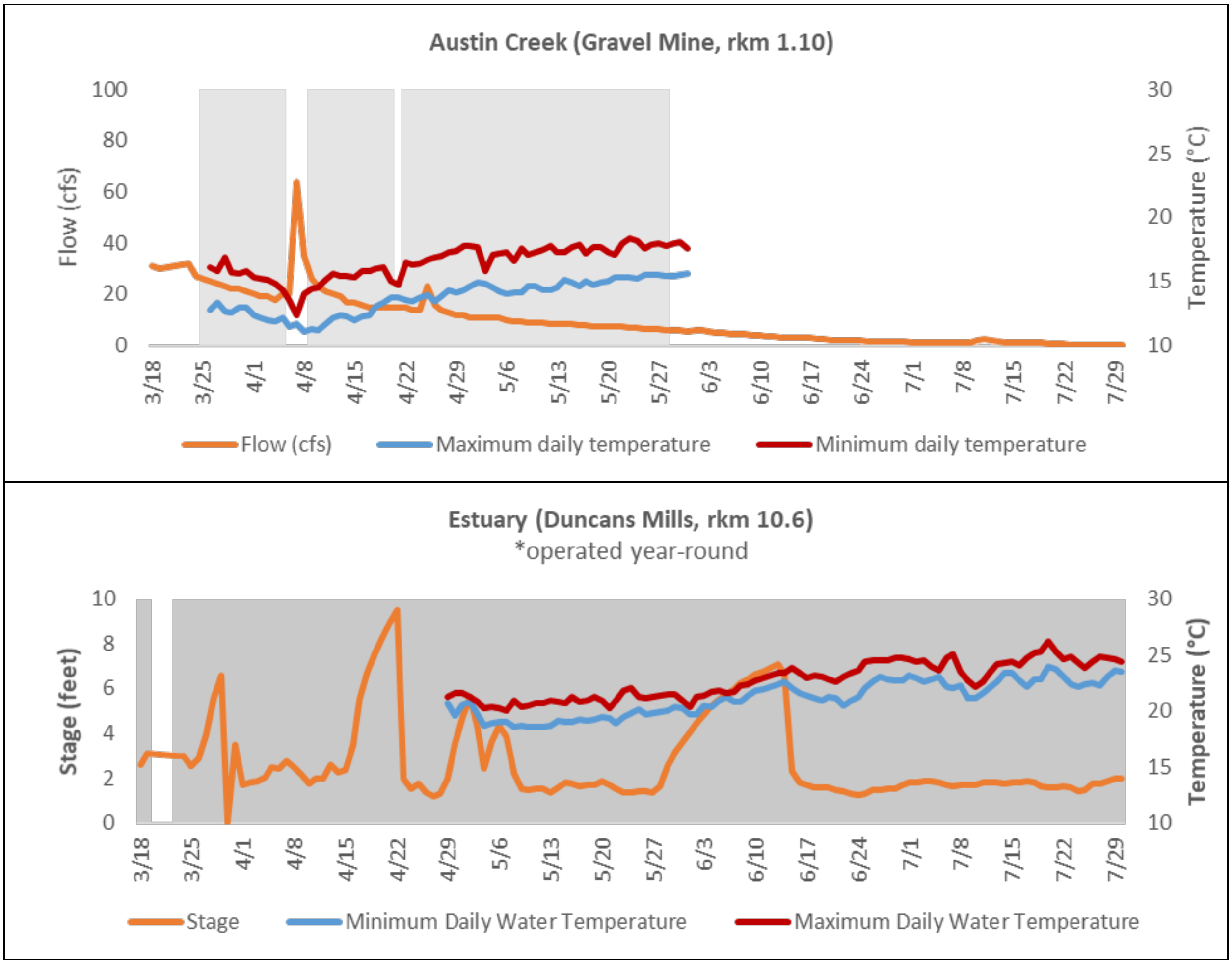


Figure 4.5.5. Environmental conditions at downstream migrant detection sites from March 18 to July 30, 2015. Gray shading indicates the proportion of each day that each facility was operated. Discharge data are from the USGS gage at Hacienda (mainstem Russian, 11467000), the USGS gage at Trenton-Healdsburg Road (Mark West Creek, 11466800), a gage operated by CEMAR on Dutch Bill Creek (data unavailable in 2015) and the USGS gauge at Cazadero (Austin Creek, 11467200). Stage data for the estuary are from the Jenner gage. Temperature data are from the data loggers operated by the Water Agency at each monitoring site.

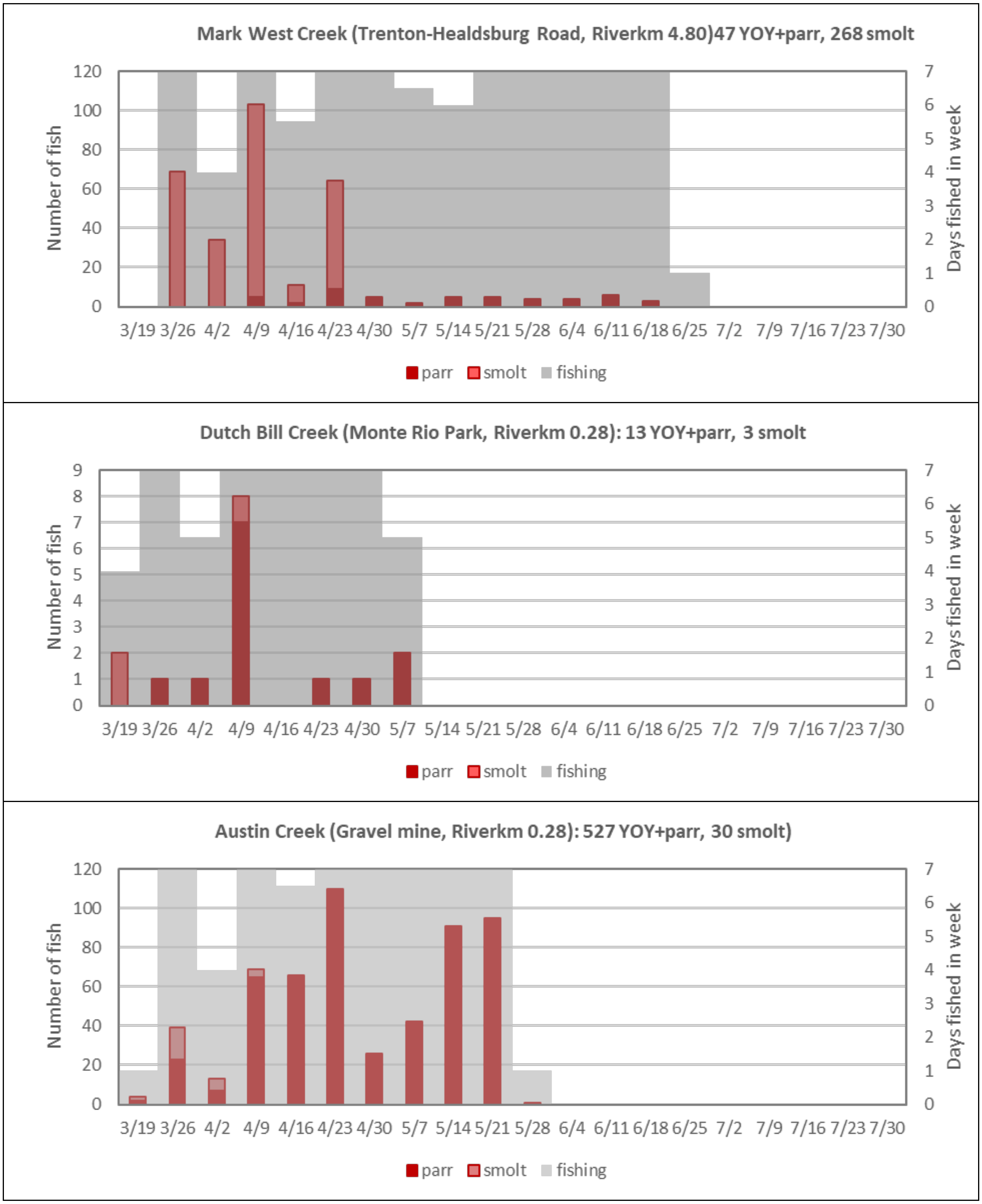


Figure 4.5.6. Weekly capture of steelhead by life stage at lower river downstream migrant trapping sites, 2015. Gray shading indicates portion of each week trap was fishing. Note the different vertical scale among plots for each site.

Table 4.5.3. Number of steelhead juveniles PIT-tagged at downstream migrant traps, 2009-2015.

Site	2009	2010	2011	2012	2013	2014	2015
Dry Creek	no tagging	no tagging	no tagging	no tagging	2,703	1,348	No tagging
Mainstem	5	96	99	315	100	101	not fished
Mark West Creek	not fished	not fished	not fished	43	135	18	19
Dutch Bill Creek	not fished	46	22	6	12	21	7
Austin Creek	not fished	996	500	1636	1749	590	107
Total	**5**	**1,138**	**621**	**2,000**	**4,699**	**2,078**	**133**

Of the 133 juvenile steelhead that were PIT-tagged in downstream migrant traps in 2015, 38 (28.6%) were detected on the PIT antenna array at Duncans Mills (Table 4.5.4) and 35 of those 38 were PIT-tagged at Austin Creek. Reasons for non-detection include an unknown number of fish that simply did not move into the estuary as well as fish that moved into the tidal portion of the estuary but were not detected due to imperfect PIT antenna array detection efficiency at Duncans Mills.

Fewer steelhead were captured and available for tagging at Austin Creek than in all previous years. Over the course of the season, 557 steelhead were captured of which 511 were YOY. A total of 107 juvenile steelhead were PIT-tagged; however, based on their size, only 42 of these PIT tagged fish were YOY (Figure 4.5.7). In total, 101 PIT-tagged steelhead were released upstream of the trap and 6 were released downstream of the trap (Table 4.5.5). Because 31 of the 42 PIT-tagged YOY were detected on the PIT antenna array just downstream of the trap in Austin Creek, we have high certainty that at least 73.8% (31/42) moved downstream into the estuary/lagoon. Because of imperfect antenna detection efficiency, we expanded those minimum counts that were based only on PIT-tagged YOY as follows.

Of the 35 PIT-tagged steelhead from Austin Creek (YOY and parr) detected on the downstream antenna in the array (Duncans Mills), 34 were also detected on the upstream antenna array (Austin Creek) resulting in an estimated antenna efficiency of 97.1 % (34/35). In order to estimate the number of YOY out of the original 42 that actually moved downstream of the Austin antenna array, we used this proportion to expand the 31 detections to 32 (31/97.1%).

Of the YOY detected on either the downstream PIT antenna arrays that were also released upstream of the trap, none were recaptured in the trap resulting in a trap efficiency that is too low to estimate. Because recapture numbers were so low, we are unable to estimate the population size of steelhead YOY moving past the trap site which meant we also could not estimate the number of YOY that emigrated to the estuary.

Figure 4.5.4. Number of steelhead captured at downstream migrant traps, number PIT tagged and number detected on the Duncans Mills PIT tag detection system prior to October 15, 2015.

Site	Number Captured	Number PIT-Tagged	Number (proportion) Detected at Duncans Mills
Mainstem	not fished	not fished	0 (0.0%)
Mark West Creek	315	19	0 (0.0%)
Dutch Bill Creek	16	7	3 (42.9%)
Austin Creek	557	107	35 (34.6%)
Total	**888**	**133**	**38 (28.6%)**

When compared to Austin and Dry Creeks fewer numbers of juvenile steelhead were captured at Mark West and Dutch Bill Creeks (Figure 4.5.6) meaning that fewer numbers of juvenile steelhead were PIT-tagged at these locations (Table 4.5.4). Fork lengths of fish caught at these traps show at least 3 year classes with steelhead YOY present at each of the trapping locations (Figure 4.5.7). As in other years, we assume that the few steelhead smolts captured at any of the trap sites was likely due to a large portion of the smolt outmigration occurring before trap installation and the generally low trap efficiencies for steelhead smolts that is well-documented in the Russian River and elsewhere. The season total catches of steelhead have been variable over the course of years monitored (Figure 4.5.8 through Figure 4.5.10).

Coho

At Mark West Creek, 884 hatchery smolts, 80 wild smolts, and 87 smolts of unknown origin were detected at the trap (Figure 4.5.8 and Figure 4.5.11). A total of 179 hatchery and 8 wild coho smolts, 14 smolt of unknown origin were captured at the Dutch Bill Creek trap (Figure 4.5.9 and Figure 4.5.11). At Austin Creek, 72 hatchery smolts, 9 wild smolts, 34 smolts of unknown origin, and 9 wild parr were captured (Figure 4.5.10 and Figure 4.5.11). Based on length data collected at the lower river traps, there were at least two age groups (YOY: age-0 and parr/smolt: ≥age-1) of coho captured (Figure 4.5.12). For a more detailed analysis of downstream migrant trapping catches of coho from other Russian River streams see UCCE Coho Salmon Monitoring Program results for 2015.

Chinook

In 2015 relatively few Chinook smolts were captured in Austin Creek, Dutch Bill Creek, and Mark West Creek (164, 0, and 1,341, respectively). For more details on characteristics of Chinook smolts captured at Dry Creek see the Russian River Biological Opinion Status and Data Report year 2015-2016.

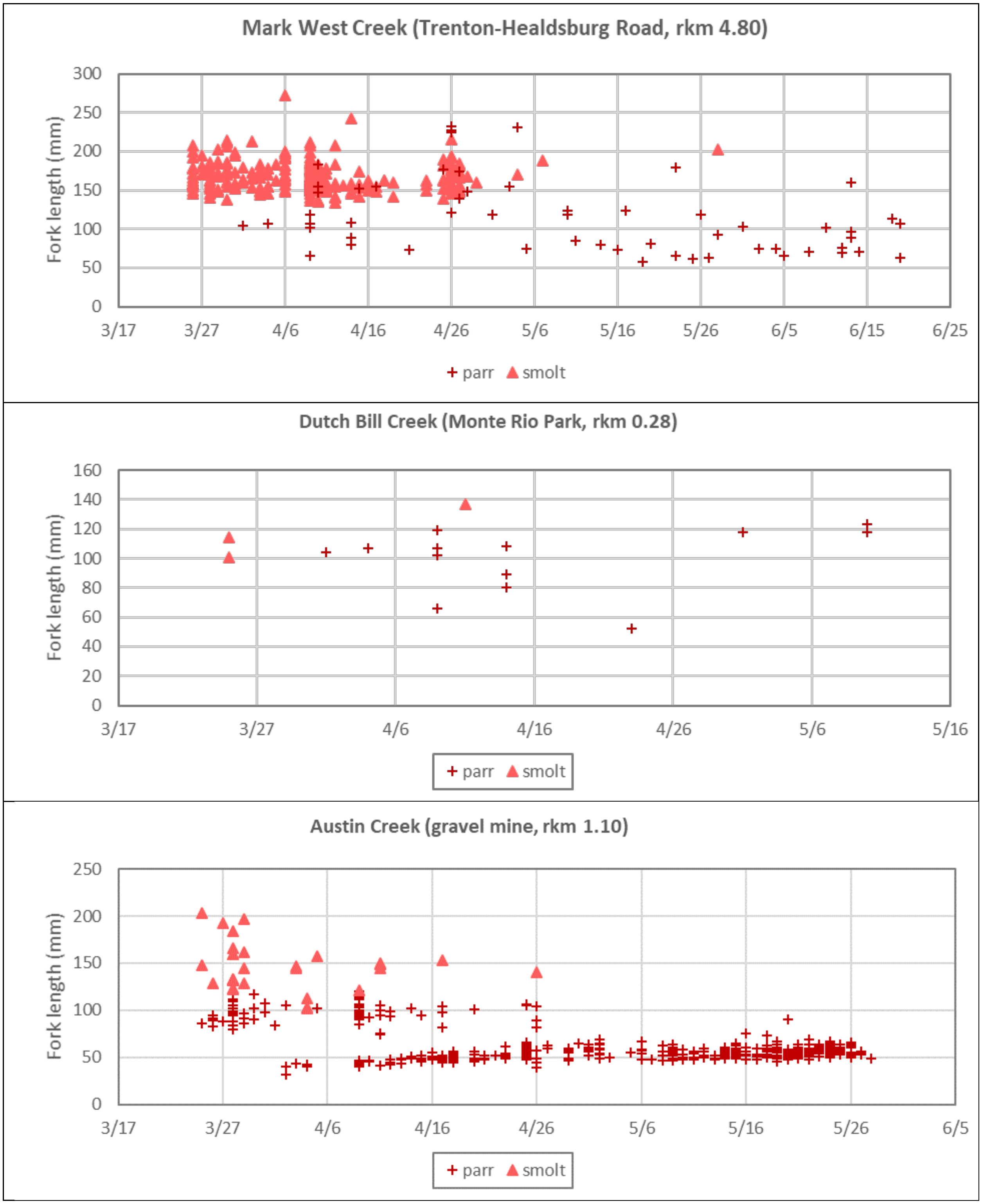


Figure 4.5.7. Weekly fork lengths of juvenile steelhead captured at lower river downstream migrant trap sites, 2015

Table 4.5.5. PIT tag and trap capture metrics and values for YOY steelhead in Austin Creek. Note that 2010 numbers differ from Manning and Martini-Lamb (2011) because they have been adjusted to only include YOY.

Metric	2010	2011	2012	2013	2014	2015
Number PIT-tagged YOY released upstream of trap	765	324	1,356	0	214	42
Number PIT-tagged YOY released downstream of trap	195	2	162	1,746	269	6
Number PIT-tagged YOY detected on antenna array that were tagged in Austin Creek	547	131	574	1,335	275	13
Number PIT-tagged YOY released upstream & detected on antenna array	389	131	486	0	57	13
Number released upstream & recaptured in trap & detected on antenna	47	8	196	0	2	0
ESTIMATED TRAP EFFICIENCY	**12.1%**	**6.1%**	**40.3%**	**N/A**	**N/A**	**N/A**
Number YOY+parr detected on both antennas in array	241	93	85	399	129	34
Number YOY+parr detected on downstream antenna only	288	178	129	463	162	35
ESTIMATED ANTENNA EFFICIENCY	**83.6%**	**52.2%**	**65.9%**[1]	**86.2%**[1]	**79.6%**[1]	**97.1%**
Number YOY captured and PIT-tagged (≥60 mm only)	960	324	1,518	1,746	483	42
Total number of YOY captured (≥60 mm only)	2,617	453	2,341	4,216	541	42
ESTIMATED NUMBER OF PIT-TAGGED YOY EMIGRANTS (≥60 mm only)	**632**	**251**	**759**	**1,549**	**325**	**32**
ESTIMATED PROPORTION OF PIT-TAGGED YOY THAT EMIGRATED (≥60 mm only)	**65.8%**	**77.5%**	**50%**	**88.5%**	**67.3%**	**76.2%**
ESTIMATED POPULATION SIZE OF YOY AT TRAP	**21,628**	**7,426**	**5,804**	**N/A**	**N/A**	**N/A**
ESTIMATED NUMBER OF YOY IN POPULATION THAT EMIGRATED	**14,231**	**5,755**	**2,901**	**N/A**	**N/A**	**N/A**

[1]Efficiency is based on detections of PIT-tagged fish at Duncans Mills.

Figure 4.5.8. Number of steelhead and coho salmon captured by life stage and origin at the Mark West Creek downstream migrant trap, (upper panels) and duration and timing of trap operation (lower panel), 2009-2015.

Figure 4.5.9. Number of steelhead and coho salmon captured by life stage and origin at the Dutch Bill Creek downstream migrant trap, (upper panels) and duration and timing of trap operation (lower panel), 2009-2015.

Figure 4.5.10. Number of steelhead and coho salmon captured by life stage and origin at the Austin Creek downstream migrant trap, (upper panels) and duration and timing of trap operation (lower panel), 2009-2015.

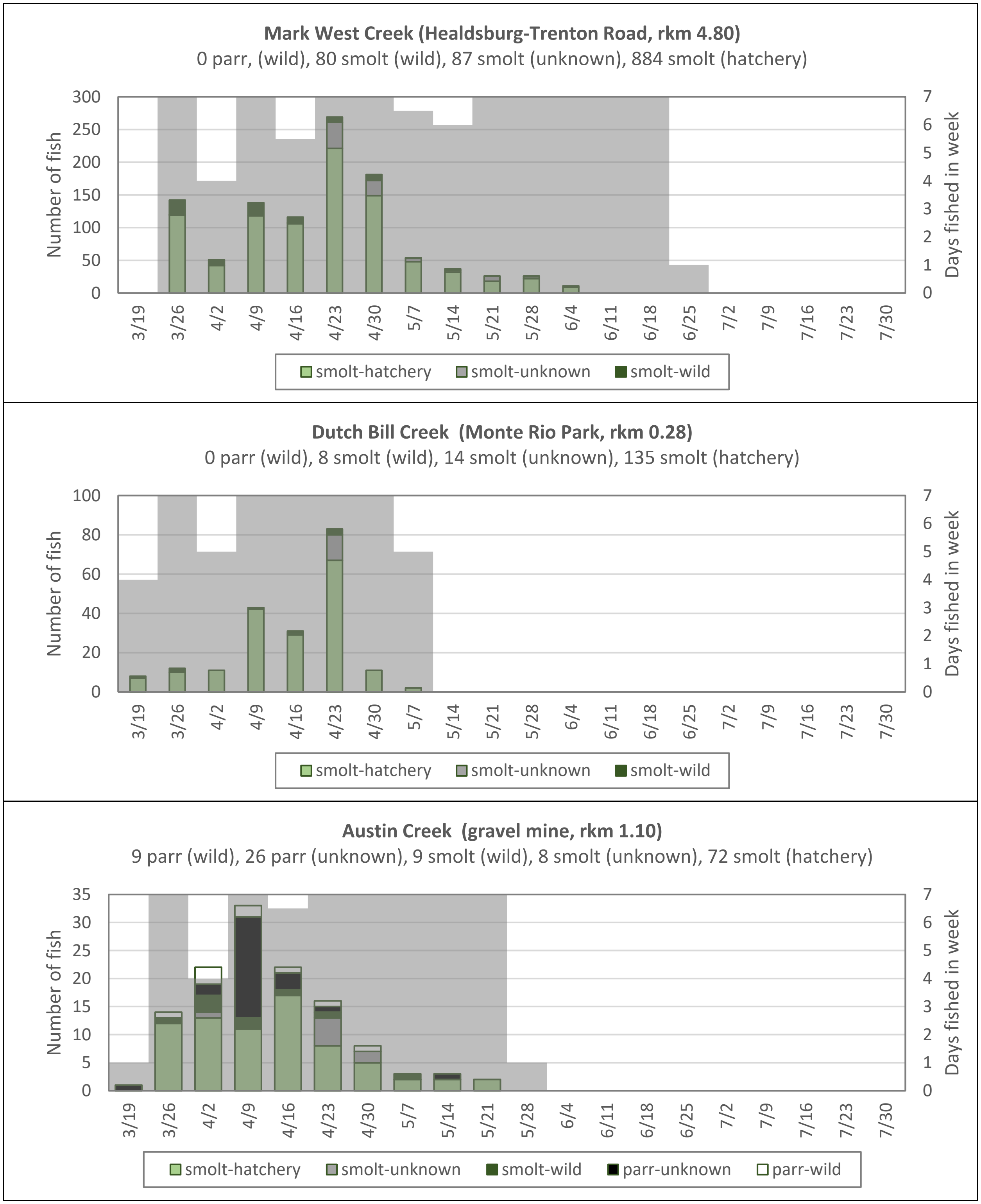


Figure 4.5.11. Weekly capture of coho salmon by life stage at lower river downstream migrant trapping sites, 2015. Gray shading indicates portion of each week trap was fishing. Note the different vertical scale among plots for each site.

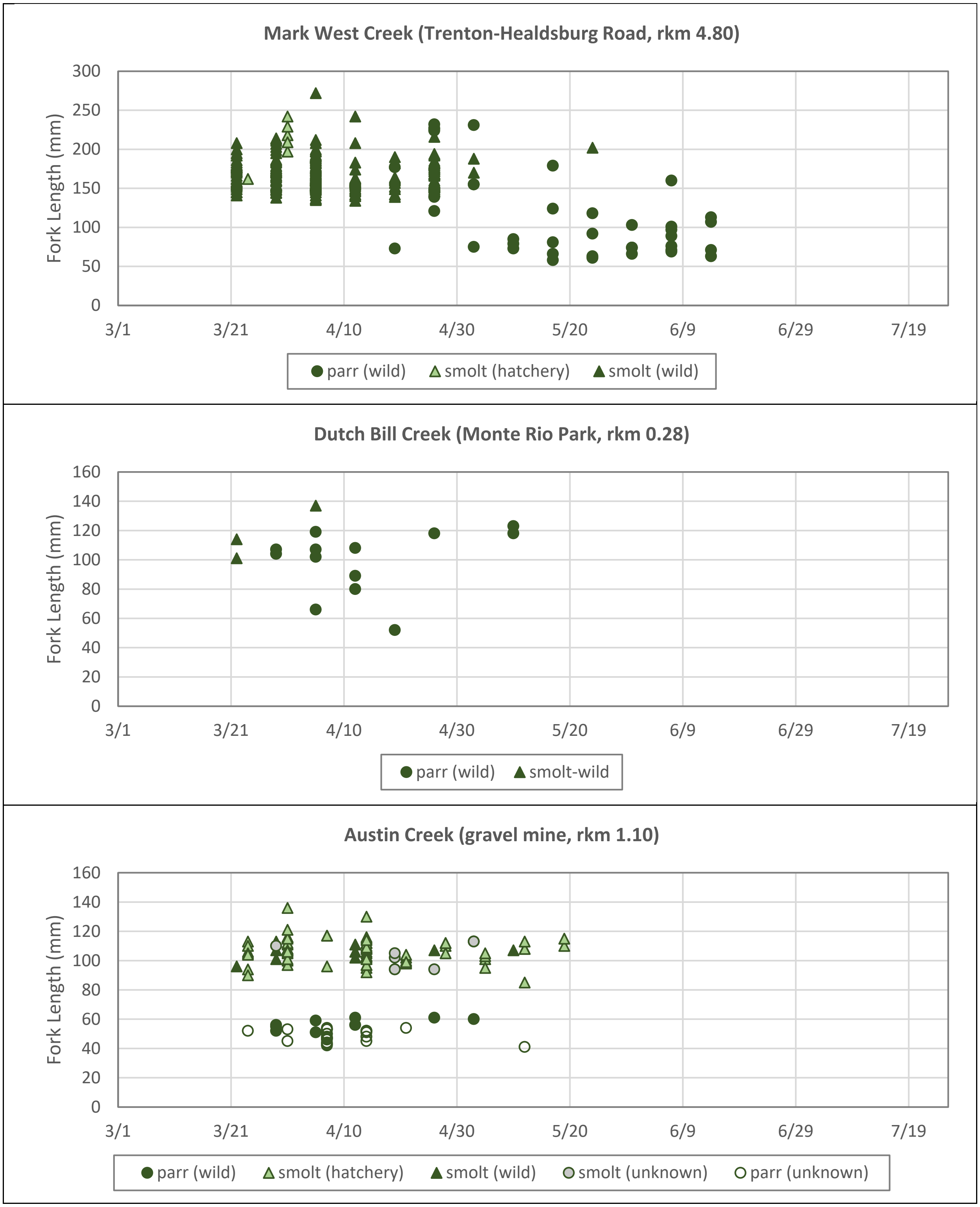


Figure 4.5.12. Weekly fork lengths of coho salmon captured at lower river downstream migrant trap sites, 2015.

Conclusions and Recommendations

Russian River Biological Opinion objectives regarding the timing of estuary entry are partially met by using PIT tag detections from the paired antenna array in lower Austin Creek where antenna efficiency estimates are possible and where fish moving past that array have effectively entered the estuary. In past years many steelhead YOY were detected leaving Austin Creek and entering the estuary. This same pattern was not seen at the other tributary monitoring site. In 2015, low trap efficiency caused multiple problems in our ability to monitor steelhead at Austin Creek. Because trap efficiency was low few steelhead were captured and available for tagging. The low number of tagged fish led to fewer opportunities to detect fish on PIT tag antennas as they entered the estuary or during seining surveys after these fish took residence in the estuary. Low trap efficiency at Austin Creek precluded estimating the population of Austin Creek YOY migrating past the trap and the population that entered the estuary. It is likely that the recent drought impacted the number of juvenile steelhead emigrating from Austin Creek. However, because trap efficiency was low the actual number of steelhead that entered the estuary was likely higher than the number of steelhead encountered at the trap in 2015.

While the PIT tag antenna at Duncans Mills spanned the Russian River for the 2015 outmigration season, detections of PIT tagged fish were not guaranteed because there are sections between antennas where fish could pass undetected. Fish orientation, and multiple PIT-tagged fish in the detection field of the same antenna at the same time can also effect detection probability. Brackish water occasionally occurs at the antenna site which cause decreases in antenna read range and water depths may exceed the detection field of some antennas. Collectively, these limitations all result in decreases in overall antenna efficiency; however, they are non-issues as long as detection efficiency can be estimated for use in expanding the number of fish detected. Unfortunately, efficiency estimates at Duncans Mills have not been possible because of the lack of a second antenna array in close proximity to the first (e.g., as is the case in Austin Creek, Figure 4.5.4). Regardless of these issues, PIT-tagging steelhead YOY at upstream locations and detecting those individuals if and when they move into the estuary (along with beach seining in the estuary itself) remain as the only viable method we know of for addressing the fish monitoring objectives in the Russian River Biological Opinion. Attempts continue to measure antenna efficiency so that expanded counts of PIT tagged individuals passing the antenna array can be constructed in future years.

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CHAPTER 5 : Dry Creek Habitat Enhancement, Planning, and Monitoring

Dry Creek Habitat Enhancement

The Biological Opinion contains an explicit timeline that prescribes a series of projects to improve summer and winter rearing habitat for juvenile coho salmon and steelhead in Dry Creek (Figure 5.1.1). During the initial three years of implementation, 2008 to 2011, the Water Agency is charged with improving fish passage and habitat in selected tributaries to Dry Creek and the lower Russian River. The status of those efforts is described in Chapter 6 of this report. For the mainstem of Dry Creek, during this initial period, the Water Agency was directed to perform fisheries monitoring, develop a detailed adaptive management plan, and conduct feasibility studies for large-scale habitat enhancement and a potential water supply bypass pipeline. The pipeline feasibility study was completed in 2011 and is reported in Martini-Lamb and Manning 2011.

In 2012, the Water Agency began construction of the first phase of the Dry Creek Habitat Enhancement Demonstration Project. A second phase of the Dry Creek Habitat Enhancement Demonstration Project was constructed in 2013 with a third and final phase scheduled for construction in 2014. The Dry Creek Habitat Enhancement Demonstration Project consists of a variety of habitat enhancement projects along a section of Dry Creek a little over one mile in length in the area centered around Lambert Bridge. Concurrently, the U.S. Army Corps of Engineers completed construction in 2013 of a habitat enhancement project on U.S. Army Corps of Engineers owned property just below Warm Springs Dam (Reach 15 area).

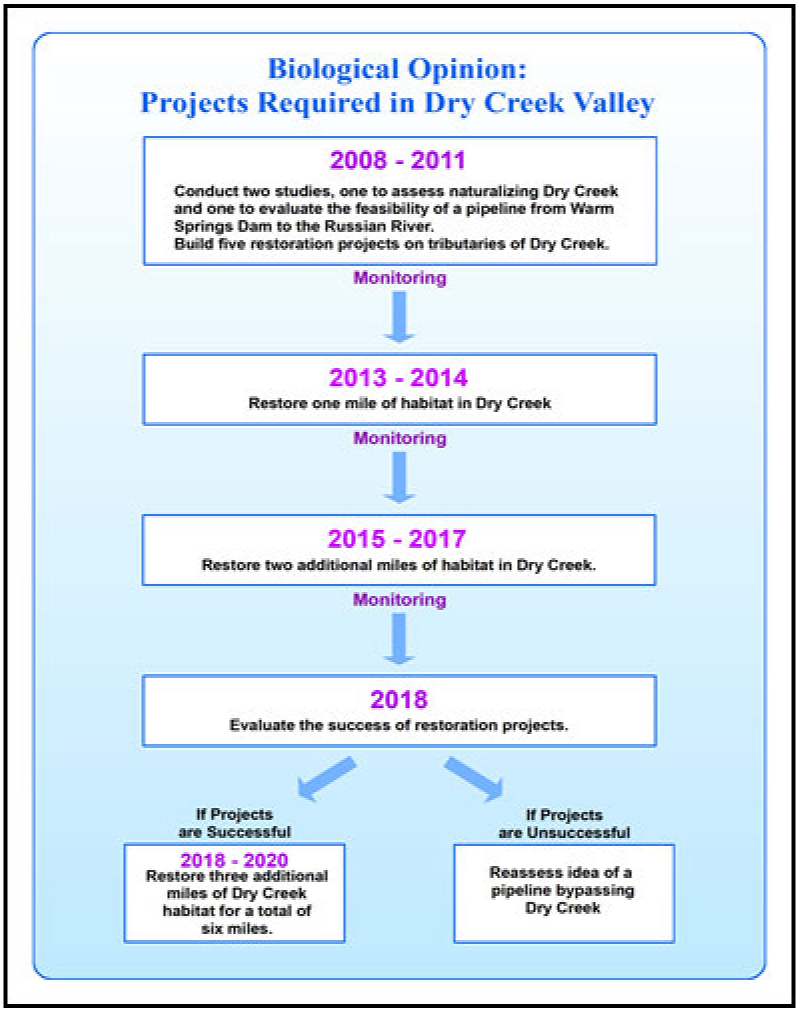

Figure 5.1.1. Timeline for implementation of Biological Opinion projects on Dry Creek.

Habitat Enhancement Feasibility Study

The Water Agency regulates summer releases from Warms Springs Dam along a 14 mile reach of Dry Creek from Lake Sonoma to the Russian River. This abundant, cool, high quality water has tremendous potential to enhance the Russian River's coho and steelhead population but it flows too swiftly to provide maximum habitat benefit. By modifying habitat conditions to create refugia from high water velocities along 6 miles of Dry Creek, NMFS and DFG assert that water supply releases can continue at current discharge levels of approximately 100 cubic feet per second (cfs) and potentially historic discharge levels up to 175 cfs.

To plan large scale enhancement of the Dry Creek channel, the Water Agency has retained Inter-Fluve, Inc. to conduct extensive field surveys and produce a series of reports detailing habitat enhancement opportunities along Dry Creek. Interfluve's work is being conducted in three phases: 1) inventory and assessment of current conditions; 2) feasibility assessment of habitat improvement approaches; and 3) conceptual design of habitat approaches deemed feasible. All three reports have been completed and can be viewed at http://www.scwa.ca.gov/drycreek/.

During 2011, Interfluve developed the Dry Creek Fish Habitat Enhancement Conceptual Design Report (Appendix 5.1). The final report was released to the public in July 2012 and identifies 26 sub reaches along Dry Creek as potential areas for construction of low velocity habitat with depth and cover characteristics conducive to rearing juvenile coho salmon and steelhead.   The opportunities identified in the report are distributed throughout the 14 mile length of Dry Creek. However, different reaches of Dry Creek present unique geomorphic and hydrologic constraints and Interfluve divided the stream into upper, middle, and lower segments. In the upper segment (mile 11 to 13.7), the influence of Warm Springs Dam on streamflow, substrate, and channel dimensions is most pronounced. The stability of this reach provides opportunities for long lasting “constructed” habitat features such as side channels, backwaters, and log structures. In the lower segment between Westside Road Bridge and the confluence with the Russian River (mile 0 to 3), conditions are amenable to constructing projects designed to let natural river processes develop habitat over time. The middle segment between Pena Creek and Westside Road (mile 3 to 11), has opportunities for both constructed habitat and river process based approaches.

The Concept Design report includes a description of current habitat conditions, modeled inundations at high flow, maps and graphics depicted proposed summer and winter habitat features, and a preliminary cost estimate for each of the 26 enhancement sub reaches along Dry Creek (Figure 5.1.2). All of the sub reaches are ranked according to the potential quantity of summer and winter coho rearing habitat they provide (Table 5.1.1). This ranking does not, however, include implementation considerations such as relative cost, landowner willingness and accessibility, and continuity or predicted longevity of constructed features. Figure 5.1.3 illustrates the two step process that will be employed to select enhancement reaches on Dry Creek.

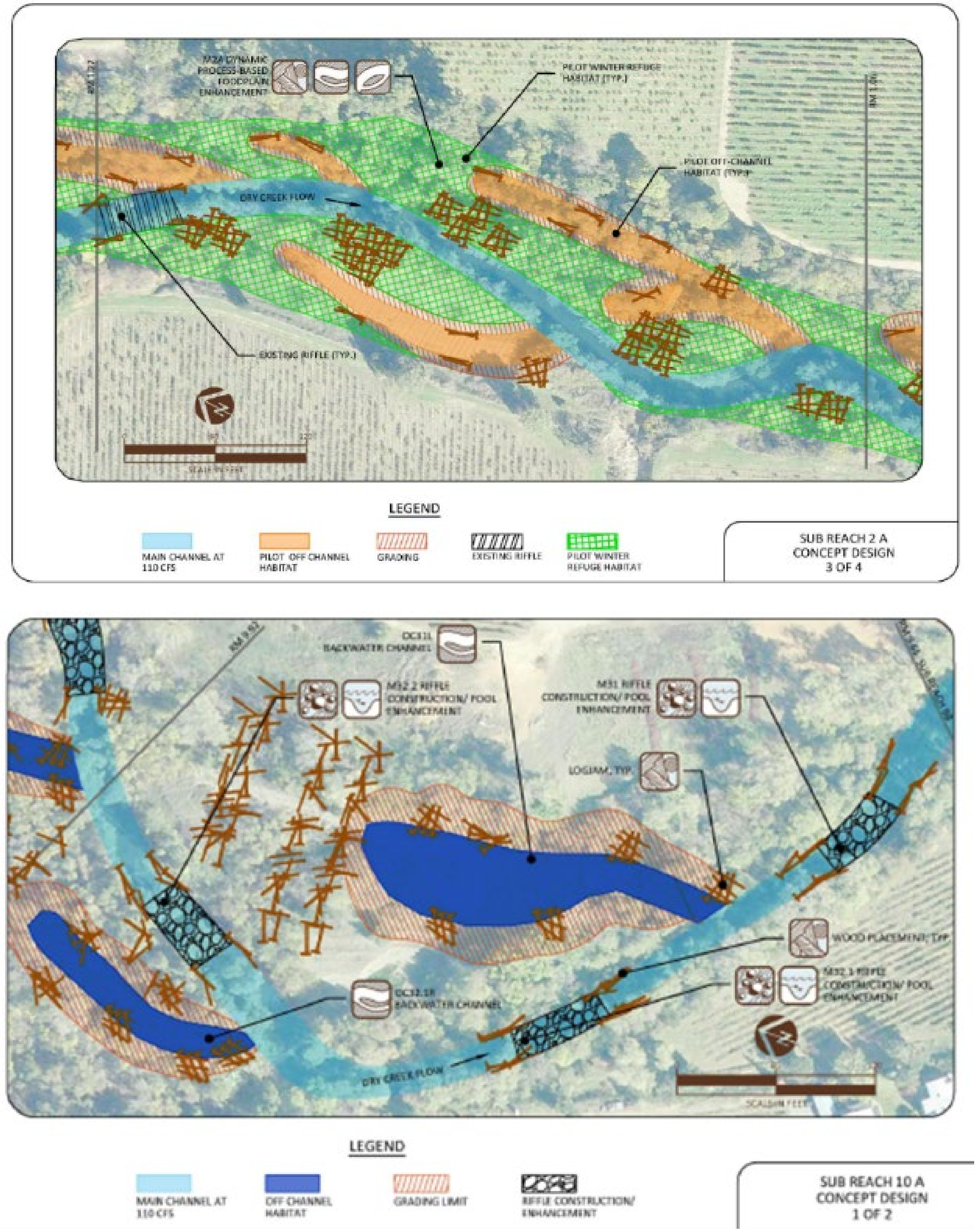

Figure 5.1.2. Examples of habitat enhancement conceptual designs for two Dry Creek subreaches. The top panel, Reach 10A, illustrates proposed summer habitat enhancements using a static "constructed" habitat approach. Reach 2A, lower panel, is close the confluence of Dry Creek and the mainstem Russian River. In this highly dynamic environment, a "process" based approach that creates pilot habitat features the stream can adjust over time is proposed.

Table 5.1.1. Ranking of enhancement subreaches in Dry Creek organized by Upper, Middle, and Lower segments.

Segment	Ranking Tier	(Sub) Reach	Coho Potential Coho Rearing Habitat Score	Winter Refuge & Rearing Habitat Score	Total Potential Habitat Score	Predicted Continuity Score
Upper	Tier I	14a	High	Medium	High	High
Upper	Tier I	13b	Medium	Medium	Medium	High
Upper	Tier I	15	Medium	Low	Low	High
Upper	Tier I	14b	Medium	Low	Low	High
Upper	Tier II	12b	Low	High	Medium	High
Upper	Tier II	13a	Low	Low	Low	High
Upper	Tier II	12a	Low	Low	Low	High
Middle	Tier I	8b	High	Medium	High	Medium
Middle	Tier I	4a	High	Low	High	High
Middle	Tier I	5a	High	Low	High	Medium
Middle	Tier I	4b	High	Low	Medium	Medium
Middle	Tier I	8a	Medium	High	High	High
Middle	Tier I	5b	Medium	Medium	High	Medium
Middle	Tier I	10a	Medium	Low	Medium	High
Middle	Tier I	10b	Medium	Low	Medium	Medium
Middle	Tier I	4c	Medium	Low	Low	High
Middle	Tier II	6	Low	High	High	Medium
Middle	Tier II	11	Low	Medium	High	Medium
Middle	Tier II	9b	Low	Medium	Low	Medium
Middle	Tier II	9a	Low	Low	Low	Medium
Lower	Tier I	2b	High	High	High	Low
Lower	Tier I	2a	High	High	High	Low
Lower	Tier I	1	High	High	High	Low
Lower	Tier II	3b	Medium	Low	Medium	Medium
Lower	Tier II	3a	Medium	Low	Medium	Medium

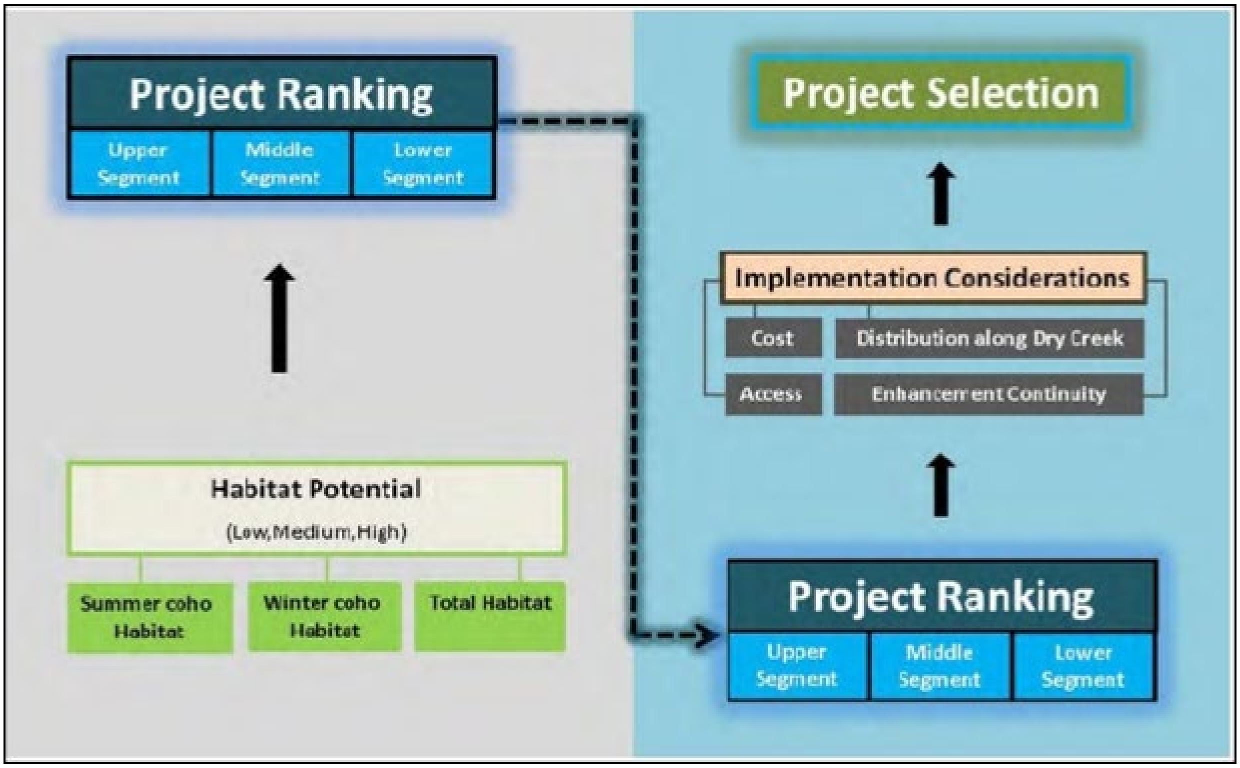

Figure 5.1.3. Conceptual depiction of habitat project prioritization approach. The left side of the figure represents the first phase of the prioritization process which includes ranking of the enhancement subreaches based solely on their inherent potential for habitat enhancement. The second phase, project selection, includes implementation considerations such as access, distribution, and cost.

Demonstration Project

As described in the Public Outreach Chapter of this report, the Water Agency must engage a diverse group of stakeholders to implement the Biological Opinion. Dry Creek is held almost entirely in private ownership and Water Agency staff must work in concert with landowners of more than 170 parcels to study, plan, and construct habitat enhancements. The Biological Opinion's 5 year timeline prior to construction of the first mile of habitat enhancement acknowledges this challenge and the depth of study, planning, and environmental compliance required for implementation. A forward looking group of property owners along a one mile stretch of the stream near Lambert Bridge, in the middle of Dry Creek Valley, approached the Water Agency with the opportunity to advance the schedule and demonstrate habitat enhancement techniques in their reach of the stream (Reach 7). The Water Agency welcomed this opportunity, and worked to implement the Dry Creek Habitat Enhancement Demonstration Project between 2012 and 2014. The U.S. Army Corps of Engineers also implemented a similar habitat enhancement (Reach 15 Project) on a 0.3 mile reach of Dry Creek immediately below Warms Springs Dam in 2013. A detailed summary of these two projects can be found in the 2015 Biological Opinion Annual Report.

Phase 2 and 3

Beyond the completion of the Demonstration Project (Reach 7) work and the Corps of Engineer's Reach 15 work, the Water Agency has continued to make progress on landowner access and design work for the next two miles of habitat enhancement. These next two miles have been designated as Phase 2 and 3, with each of these phases to be constructed in parts. No construction activities occurred in 2015; however, construction of Phase 2, Part 1 (Reach 8) and Phase 3, Part 1 (Reach 2) is expected to begin in June of 2016. Phase 2, Part 2 (Reach 14) and Phase 3, Part 3 (Reach 5) are expected to be constructed the following year by the Water Agency. Phase 3, Part 2 (Reach 4a) is expected to be constructed in 2017 or 2018 by the U.S. Army Corps of Engineers.

Figure 5.1.4. This figure shows the habitat enhancement projects along Dry Creek that have been completed and projects that are being designed.

Adaptive Management Plan and Monitoring

A question raised by the Biological Opinion is whether Dry Creek habitat enhancements will have the desired benefits. This question is important both for receiving credit toward the total amount of habitat enhancements set forth in the Biological Opinion (six miles) and for assessing the relative effectiveness of various habitat enhancements options. For the latter reason, the Biological Opinion states that "an adaptive management, monitoring and evaluation plan" will be developed that identifies "project goals, objectives and success criteria". ESSA Technologies Ltd. (an independent consulting firm from Vancouver Canada) facilitated the collaborative development of an adaptive management plan (AMP) for Dry Creek in an iterative process of meetings, discussions and document revision.

The goal of the Dry Creek AMP is to serve as a guide for monitoring juvenile coho salmon and steelhead populations and the habitats they live in over multiple years to detect change resulting from habitat enhancement. A series of multi-agency workshops were convened to address the following objectives:

1. Identify performance measures;
2. Develop success criteria for each performance measure;
3. Select approaches for evaluating performance measures relative to success criteria;

4. Agree on a set of decision rules for determining credit toward the total amount of habitat enhancement.

Evaluation of performance measures will be based on the results of **implementation** monitoring, **effectiveness** (habitat) monitoring, and **validation** (fish) monitoring.

For each type of monitoring, quantitative data for performance measures will be gathered using specific data collection protocols. These quantitative data will then be used to qualitatively rate whether the habitat enhancement was implemented correctly, whether it is having the desired effect on physical habitat conditions and whether juvenile coho and steelhead are benefiting from the work.

Implementation monitoring is “monitoring to determine if the habitat enhancement was done according to the approved design” (NMFS Russian River Biological Opinion 2008, pg. 266). In other words, did the contractor/builder do what they said they were going to do? Implementation monitoring will occur immediately post-construction and will serve as a check-in point to determine if all the essential elements were placed according to the design as approved by NMFS/CDFW. Based on the results of post-construction implementation monitoring, the Water Agency’s, USACE’s or other engineering techniques and approaches will be re-visited as deemed necessary.

Effectiveness monitoring is “monitoring to determine whether habitat enhancement is having the intended effect on physical habitat quality” (NMFS Russian River Biological Opinion 2008, pg. 266). This definition implies that protocols should facilitate a detailed comparison between baseline habitat quantity and quality data collected prior to any enhancement actions (pre-enhancement monitoring) and the habitat amounts/condition as measured over time after each implementation phase (post-enhancement monitoring). For example, pre-enhancement monitoring will occur prior to each enhancement phase, and post-enhancement monitoring will occur after the first geomorphically-effective flow (i.e., flow that deposits substantial sediment on the flood plain), or within 3 years following each enhancement phase, and then at minimum every 3 years until 2023, to assess the long term sustainability of all implemented habitat enhancement actions

Validation monitoring is “monitoring to determine whether habitat enhancement work is achieving the intended objective (i.e., creating habitat that is inhabited by listed salmonids and appreciably improves the production and survival of rearing steelhead and coho salmon in Dry Creek”; NMFS Russian River Biological Opinion 2008, pg. 266). Establishing the temporal component for validation monitoring (i.e., when should validation monitoring start and for how long) is challenging because of the inherent time lag between the physical habitat response and the expected biological response.

In addition to monitoring the habitat efforts over time (temporal scale), there is also a spatial scale at which data to evaluate habitat efforts are collected at the implementation, effectiveness, and validation monitoring stages. This spatial scale includes four progressively broader scales: feature, site, enhancement reach, project reach.

- Features: Individually engineered elements (e.g., large woody debris accumulation, riffle, pool, side channel, alcove, boulder cluster, etc.) that will individually or in composite make up a habitat enhancement site (see definition for Site below). Features can in some cases represent complete habitat units (see definition for Habitat Unit below), while in other cases they represent only structural components within a habitat unit (e.g., large wood placement).
- Site: One or more engineered habitat features (see definition for Features above) that have been designed to work in combination to enhance a stream reach.
- Enhancement reach: A specified collection of enhancement sites (see definition for site below) that are implemented in close proximity to one another.
- Project reach: A specified collection of enhancement reaches (see definition for Enhancement Reach above)

Mile 1 (Demonstration Project and USACE Reach 15 Project) Implementation Monitoring

See 2015 Biological Opinion Annual Report for a detailed report of the implementation monitoring conducted for the Demonstration Project and the U.S. Army Corps of Engineer's Reach 15 projects. Because no new construction activities occurred in 2015, no additional implementation monitoring has been conducted.

Mile 1 (Demonstration Project and USACE Reach 15 Project) Effectiveness Monitoring

As noted above under implementation monitoring, the first mile of habitat enhancement work in Dry Creek was completed in 2014. Soon after completion of the first mile of habitat, the region received significant rainfall events in December of 2014 and February of 2015, which resulted in geomorphically-effective flows in Dry Creek (5,770 cubic feet per second at Yoakim Bridge gage station on 12/11/14 and 2,530 cubic feet per second on 2/7/15).

Once geomorphically-effective flows occurred post-construction of the habitat features, Water Agency staff began collecting physical measurements (e.g. depths, velocities, temperature, dissolved oxygen, cover) to document the habitat characteristics in the project area. Initial field data collection efforts for the effectiveness monitoring was completed in December of 2015. Water Agency staff is now processing the field data. Initial evaluation of the field data indicates that the habitat features are performing as intended to meet the target depth (0.5-2.0 feet) and velocity (<0.5 feet per second) goals, which are the two primary metrics for the habitat features.

Photo 5.1.1. Dry Creek effectiveness monitoring 2015.

Photo 5.1.2. Dry Creek effectiveness monitoring 2015.

Photo 5.1.3. Dry Creek effectiveness monitoring 2015.

5.2 Effectiveness monitoring

Effectiveness monitoring focuses on the physical response of Dry Creek to habitat enhancements and determines "whether habitat enhancement is having the intended effect on physical habitat quality" in Dry Creek (NMFS Russian River Biological Opinion 2008, pg. 266). NMFS (2008) concluded that sub-optimal water velocity, depth and instream cover limit juvenile coho salmon and steelhead and suggested optimal values for water velocity depth, and cover as part of the Reasonable and Prudent Alternative (NMFS 2008). The Joint Monitoring Team, consisting of representatives from NMFS, CDFW, USACE, and the Water Agency, refined these values within the Dry Creek Adaptive Management Plan (Porter et al. 2014) and developed primary performance metrics linked to the optimal values of water velocity, depth, and cover by which to evaluate the effectiveness of habitat features, sites, and reaches (Table 5.2.1). The Joint Monitoring Team also identified secondary performance metrics that help determine the effectiveness of habitat enhancements to influence non-target, ancillary conditions (e.g., water temperature, dissolved oxygen concentration). The Dry Creek Adaptive Management Plan also suggested target flows to represent seasonal variation critical to each life stage (Porter et al. 2014).

Table 5.2.1. Primary and secondary performance measures from the Dry Creek Adaptive Management Plan.

Type of Performance Measure	Performance Measure	Life Stage	Spring Flow[1]	Summer Flow[2]	Winter Flow[3]
Primary	Velocity (ft/sec)	fry	0-0.5 ft/s	n/a	n/a
	Depth (ft)	fry	0.5-2.0 ft	n/a	n/a
	Velocity (ft/sec)	Summer/winter parr	0-0.5 ft/s	0-0.5 ft/s	0-0.5ft/s
	Depth (ft)	Summer/winter parr	2-4 ft	2-4 ft	2-4 ft
	Shelter value	Juvenile	≥80	≥80	≥80
	Pool: Riffle ratio	Juvenile	n/a	1:2 to 2:1	n/a
Secondary	Temperature (°C)	Juvenile	n/a	8-16° C	n/a
	Dissolved oxygen (mg/l)	Juvenile	n/a	6-10 mg/l	n/a
	Canopy (%)	Juvenile	80 %		
	Quiet water (< 0.5 ft/s) (%)	Juvenile	n/a	n/a	≥ 25%
	Off-channel access (off-ramps) (ft/sec)	Juvenile	Approx. 1.5 – 1.8 cm/s (Ucrit); Approx. 3.3 ft/s (burst speed)		
	Connectivity of habitats	Juvenile	Undefined		
	Substrate particle size (in.)	Adult	n/a	n/a	0.25-2.5 in.
	Depth (ft)	Adult	n/a	n/a	0.5-1.6 ft

[1] Target coho life stage during spring is newly-emerged feeding fry which use shallower depths than would be preferred later in the summer and winter when fish would be larger. Target spring flow (discharge within the enhancement reach) is 200 cfs (approximately double the summer "base" flow).
[2] Target summer flow is 105 cfs
[3] Target winter flow is 1000 cfs

Methods

The methods described below focus on data collection to assess the Dry Creek Habitat Enhancement Project against the primary performance measures of water depth (0.5-2 or 2-4 ft) and velocity (<0.5 ft/s), and amount of instream cover (shelter value) (Table 5.2.1). The remaining primary performance measure, pool to riffle ratio, is dependent on longer-term channel evolution in response to enhancement occurring after geomorphically effective flows and will be assessed in future monitoring reports. Monitoring project performance against secondary metrics is underway and will also be assessed in future reports. Depth, velocity, and shelter value provide a means to directly assess against primary metrics in the Dry Creek Adaptive Management Plan and against optimal habitat values suggested as part of the Reasonable and Prudent Alternative in the Russian River Biological Opinion (NMFS 2008, Porter et al. 2014).

Water depth and velocity

The Dry Creek Adaptive Management Plan (Porter et al. 2014) suggested collecting water depth and velocity at points along transects placed within constructed backwaters and main channel portions of Dry Creek, and “habitat feature mapping” near selected habitat enhancements (logjams, boulder fields). Habitat feature mapping would result in two-dimensional depictions of depth and velocity around habitat features and allow quantification of optimal habitat area adjacent to features. Upon consultation with NMFS, and through field experimentation with several mapping and survey tools (auto-level, differential global positioning system, total station), the Water Agency developed a robust habitat feature mapping method to characterize all portions of the Dry Creek channel, not just adjacent to enhancement features, obviating the need to collect cross-sectional data.

Field crews collected water depth and velocity at spatially referenced points across the streambed and banks using handheld flow meters and a total station. At each data point, we collected geographic location (latitude, longitude, elevation), and water depth and velocity by aiming the total station at a USGS topset rod fit with a survey prism and a flow meter (Figure 5.2.1). The technique allowed simultaneous collection of topographic and hydraulic data (water depth and velocity) that were highly spatially accurate and repeatable to enable comparison to future conditions, and allow collection of data across the streambed to create detailed relief maps. Field crews focused point collection on breaks in channel and bank slope and breaks in water velocity, and at a minimum collected points at the top of each bank, water’s edge (water surface elevation), toe of bank, thalweg, and at least two points in between the toe of bank and thalweg.

We processed the data within a Geographic Information System (GIS) to create detailed maps of stream topography (elevation) and hydraulic conditions (water depth and velocity) to spatially characterize habitat conditions and quantify optimal fry and juvenile habitat. The individual points were first used to create vector- (line) based representations of the stream channel, which were then smoothed to create raster (grid) based digital elevation models (DEMs). We classified hydraulic habitat conditions according to the primary metrics from Porter et al. (2014) (depth [0.5-2 ft or 2-4 ft], depending on life stage and velocity [<0.5 ft/s]) to identify the location of habitat falling within optimal depth, velocity, and depth and velocity ranges as polygons. Generating polygons within a GIS also allowed us to quantify the areas of optimal habitat.

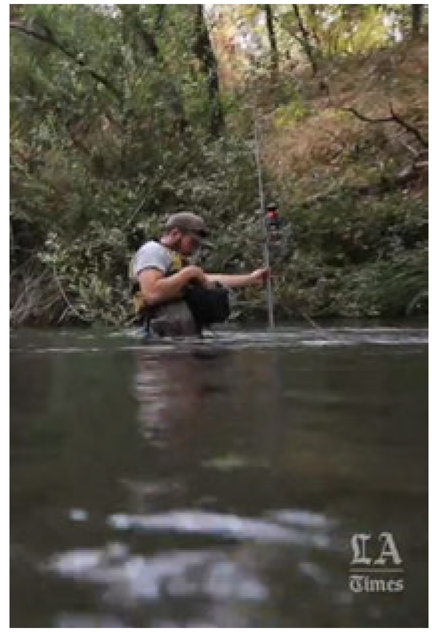

Figure 5.2.1. Dry Creek effectiveness monitoring. At each data point, we collected geographic location (latitude, longitude, elevation), and water depth and velocity by aiming the total station at a USGS topset rod fit with a survey prism and a flow meter.

Shelter value

Field crews also determined the shelter value of individual habitat units within each enhancement site. The California Salmond Habitat Restoration Manual (Flosi et al. 2010) rates instream shelter by multiplying the complexity of available cover within a habitat unit (Table 5.2.1; 0 = no shelter, 3 = highly complex shelter) by the overhead area occupied by that cover (0 = 0% of overhead area covered, 100 = 100% of overhead area covered). The maximum shelter value is 300 (3 [complexity of available cover within a habitat unit] * 100 [area of habitat unit covered]), with a score of ≥80 considered optimal within the Dry Creek Adaptive Management Plan (Table 5.2.1) (Porter et al. 2014).

We inventoried instream habitat units using habitat types described in the California Salmonid Habitat Restoration Manual (Flosi et al. 2010). These habitat types are distinguished by differences in local channel gradient, water velocity, depth, and substrate size. Flosi et al. (2010) use four hierarchical levels of classification to describe physical fish habitat, with each successive level providing greater detail. The most elementary descriptions (Levels 1 and 2) break stream channels into pool, riffle, or flatwater habitat types. Successive levels differentiate habitat types by location within the stream channel (e.g., mid-channel pools, Level 3) or by cause or agent of formation (e.g., lateral-scour, log-formed pools, Level 4). In this survey, we inventoried to habitat types to Level 2 and delineated the upstream and downstream boundaries by placing flagged 10 inch nail spikes on the right and left bank. We surveyed the location of the nail spikes with a total station and processed the data within a GIS to create polygons of habitat unit types and cover complexity.

Results

Water depth and velocity

During summer and early fall 2015, we surveyed the four enhancement reaches that make up the Demonstration Mile totaling 5,400 linear feet (223,400 ft^2) of mainstem Dry Creek, side channels and alcoves,. Field crews collected nearly 7,400 survey points, with 5,300 velocity measurements. Our analysis determined that the habitat enhancement projects created or improved approximately 91,000 ft^2 to optimal juvenile coho habitat (Table 5.2.2; Figure 5.2.2-Figure 5.2.21). The Farrow Wallace enhancement reach occupied the greatest wetted area and supported the greatest areas of optimal depth and velocity, but the second greatest amount of optimal habitat overall (Table 5.2.2, Figure 5.2.2-Figure 5.2.6). The Army Corps of Engineers enhancement reach occupied nearly half the wetted area of the Farrow Wallace enhancement reach, but created nearly as much area of optimal velocity (approximately 45,000 ft^2) and supported a greater amount of optimal habitat (Table 5.2.2, Figure 5.2.17-Figure 5.2.21). The greatest areas of optimal velocity occurred within backwaters and side channels rather than within mainstem Dry Creek (Table 5.2.2, Figure 5.2.5, Figure 5.2.10 and Figure 5.2.19). As such, the Army Corps of Engineers enhancement reach, made up of a long side channel with several lateral connections to the mainstem, but no mainstem portion, supported the greatest amount of habitat. The Rued boulder site occurred entirely within the mainstem (Figure 5.2.12-Figure 5.2.16), but created proportional habitat area (42%, [optimal habitat area/wetted area]*100) similar to the Farrow Wallace (36%) and Van Alyea (32%) enhancement reaches, which included off-channel and mainstem sites. The Rued riffle site functions as a hydraulic control to ensure inundation of a backwater feature just upstream in the Van Alyea enhancement reach and was not expected to create substantial optimal habitat.

Table 5.2.2. Wetted area, area of optimal depth and velocity, and area of optimal habitat within Dry Creek enhancement reaches in 2015.

Enhancement reach	Wetted area (ft^2)	Optimal depth (ft^2)			Optimal velocity (ft^2)	Optimal habitat (ft^2)		
		0.5 – 2.0 ft	2.0 – 4.0 ft	Total	<0.5 ft/s	0.5 – 2.0 ft < 0.5 ft/s	2.0 – 4.0 ft < 0.5 ft/s	Total
Farrow Wallace	88,000	42,372	24,300	66,672	44,874	14,596	13,235	27,831
Van Alyea	76,344	18,193	33,254	51,447	31,216	10,942	13,526	24,468
Rued (riffle site)	8,575	6,852	1,239	8,091	1,322	836	87	923
Rued (boulder site)	6,116	758	4,036	4,794	3,217	754	1,873	2,627
Army Corps	44,320	20,116	15,902	36,018	44,051	19,989	15,764	35,753
TOTAL	**223,355**	**88,291**	**78,731**	**167,022**	**124,680**	**47,117**	**44,485**	**91,602**

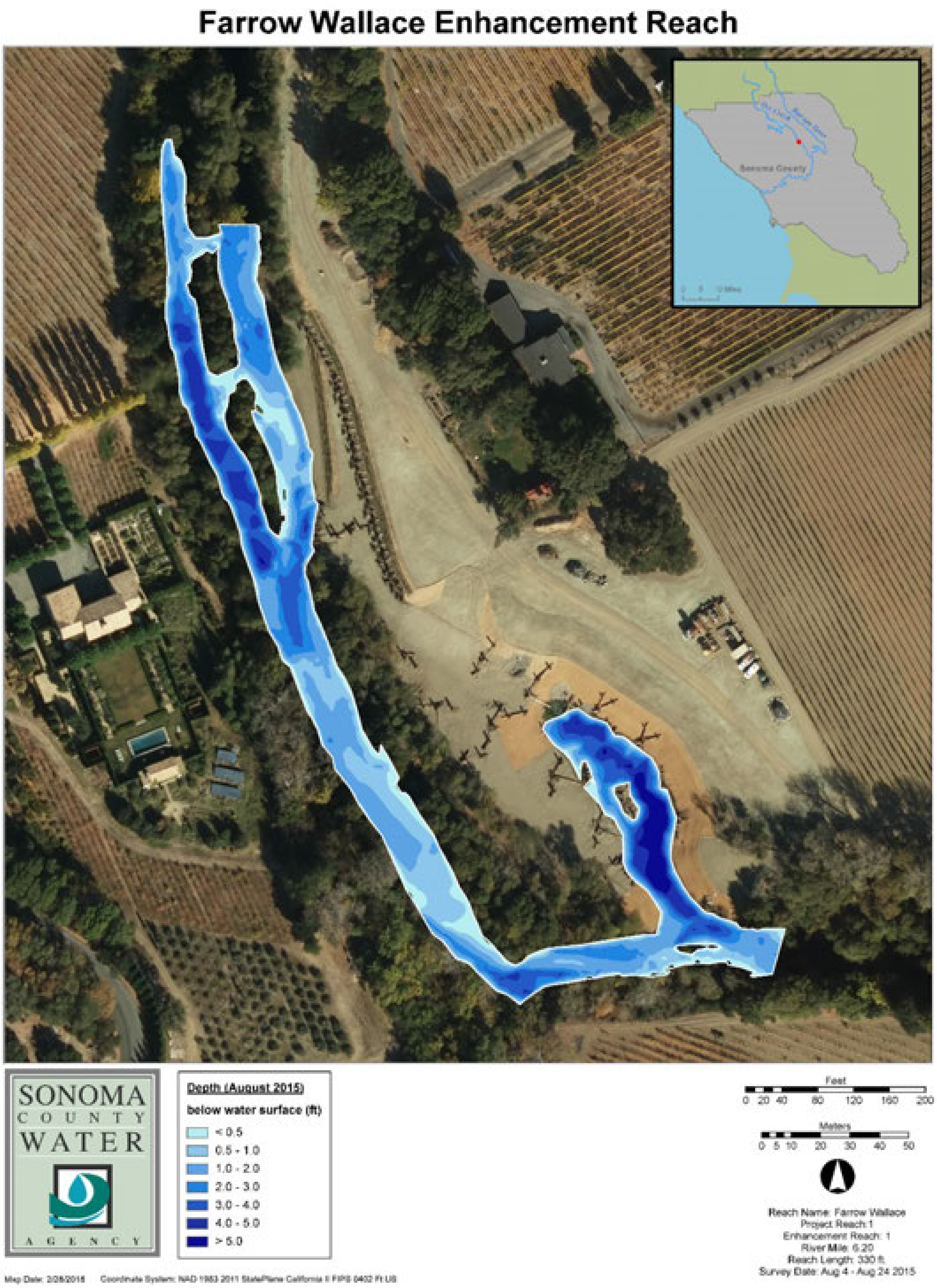

Figure 5.2.2. Measured water depth within the Farrow-Wallace habitat enhancement reach during August 2015.

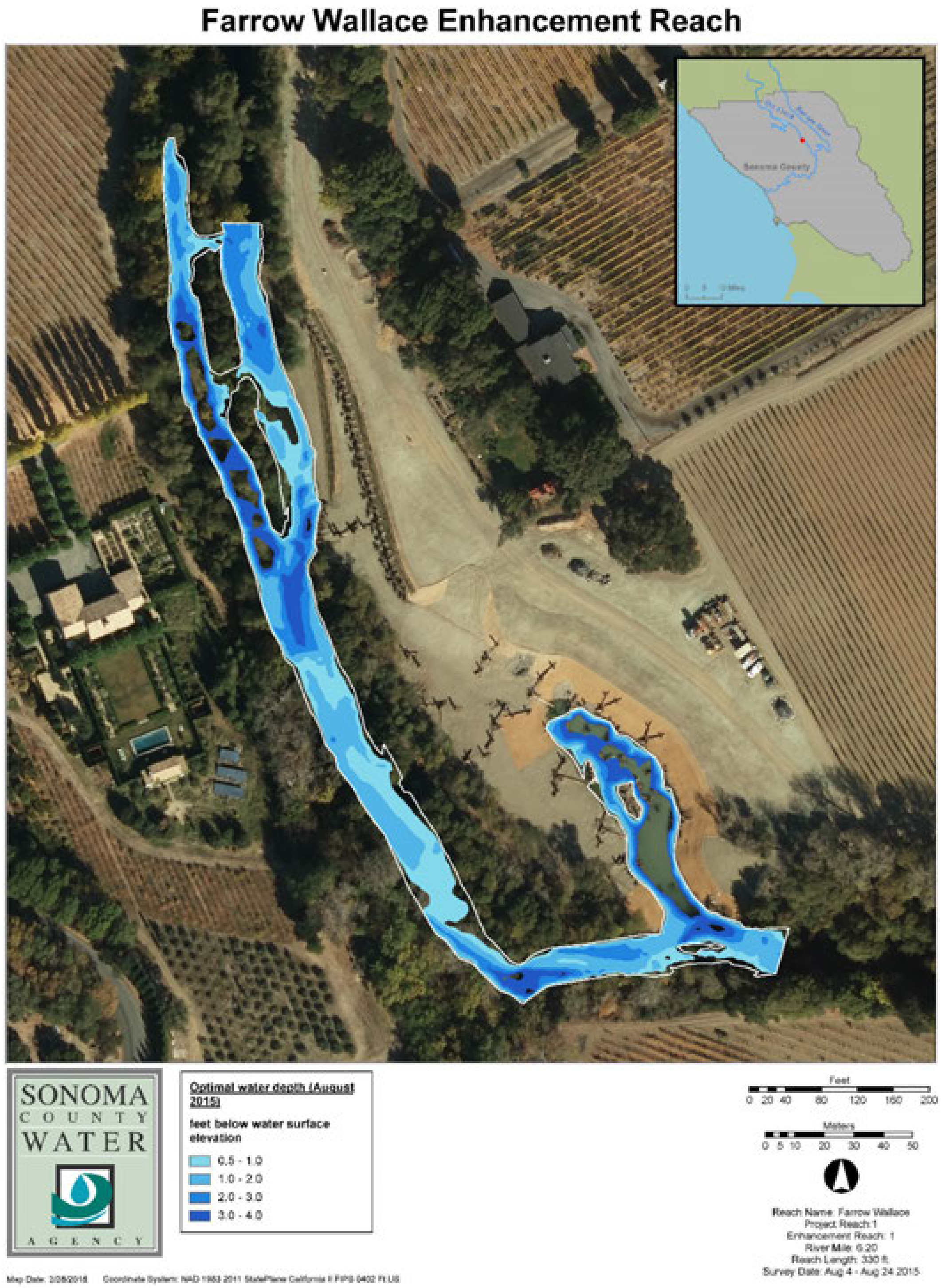

Figure 5.2.3. Area of optimal water depth within the Farrow-Wallace habitat enhancement reach during August 2015.

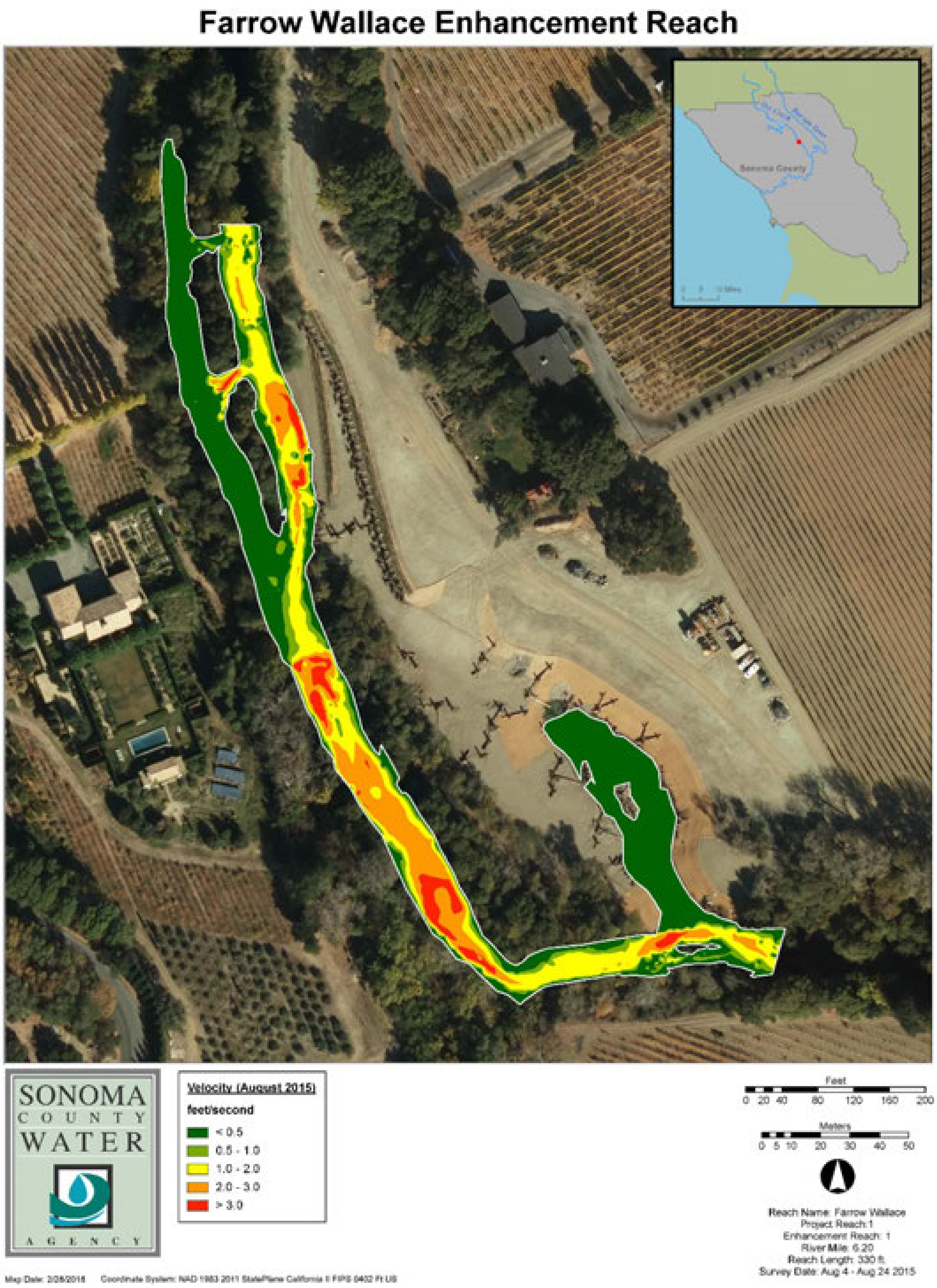

Figure 5.2.4. Measured water velocity within the Farrow-Wallace habitat enhancement reach during August 2015.

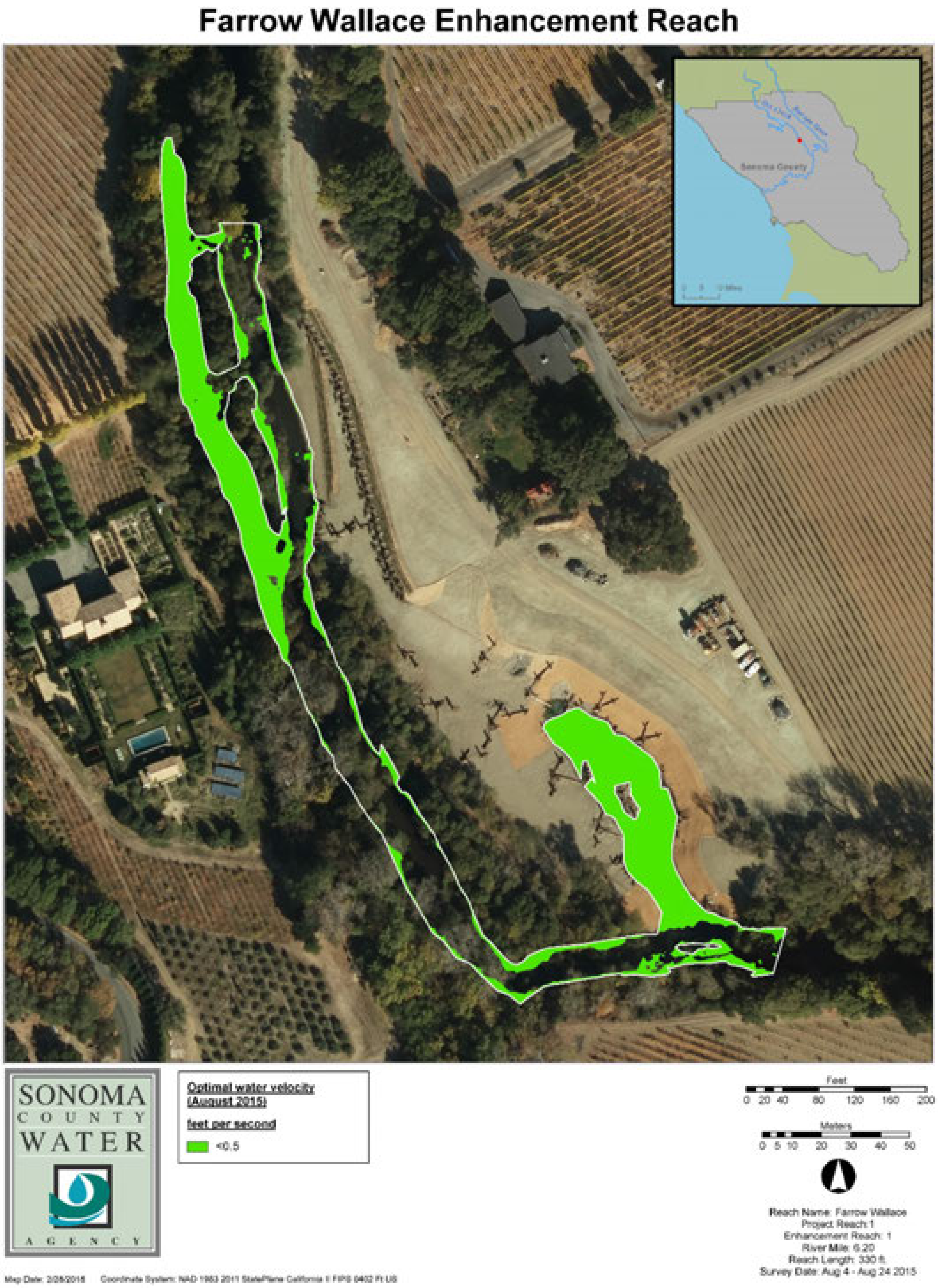

Figure 5.2.5. Area of optimal water velocity within the Farrow-Wallace habitat enhancement reach during August 2015.

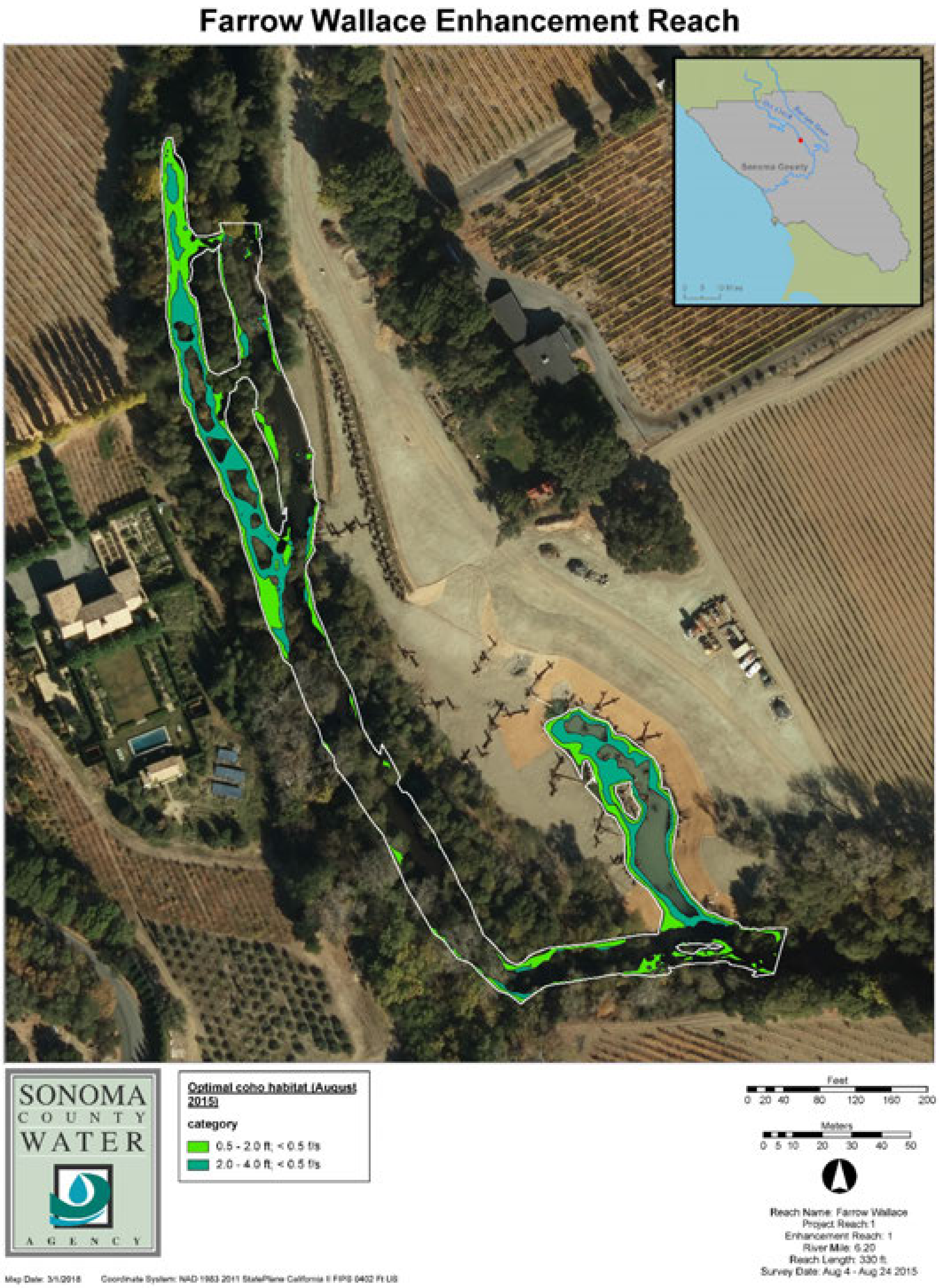

Figure 5.2.6. Area and location of optimal fry (<0.5 f/s, 0.5-2.0 ft) and parr (<0.5 f/s, 2.0-4.0 ft) habitat within the Farrow-Wallace habitat enhancement reach during August 2015.

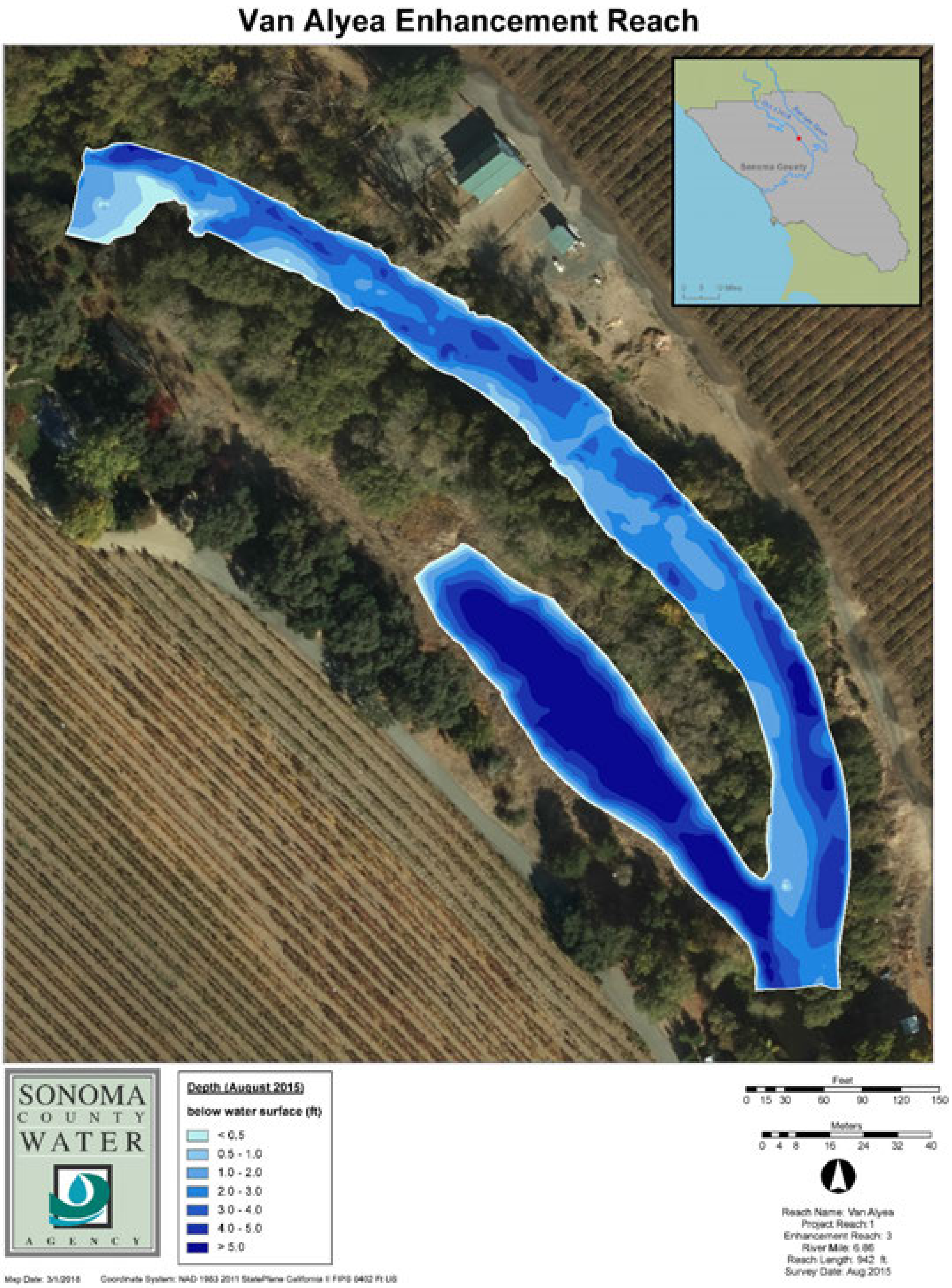

Figure 5.2.7. Measured water depth within the Van Alyea habitat enhancement reach during August 2015.

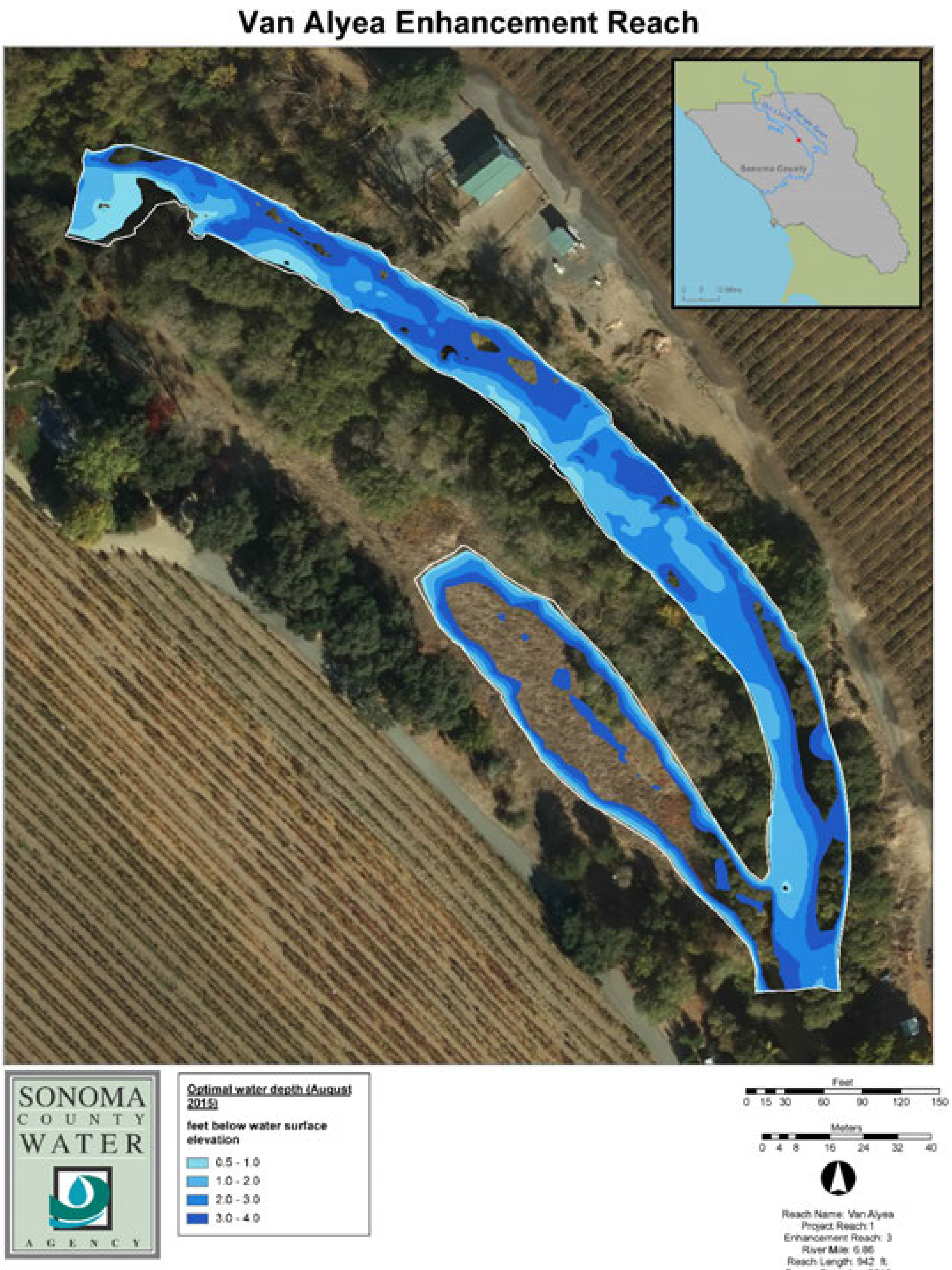

Figure 5.2.8. Area of optimal water depth within the Van Alyea habitat enhancement reach during August 2015.

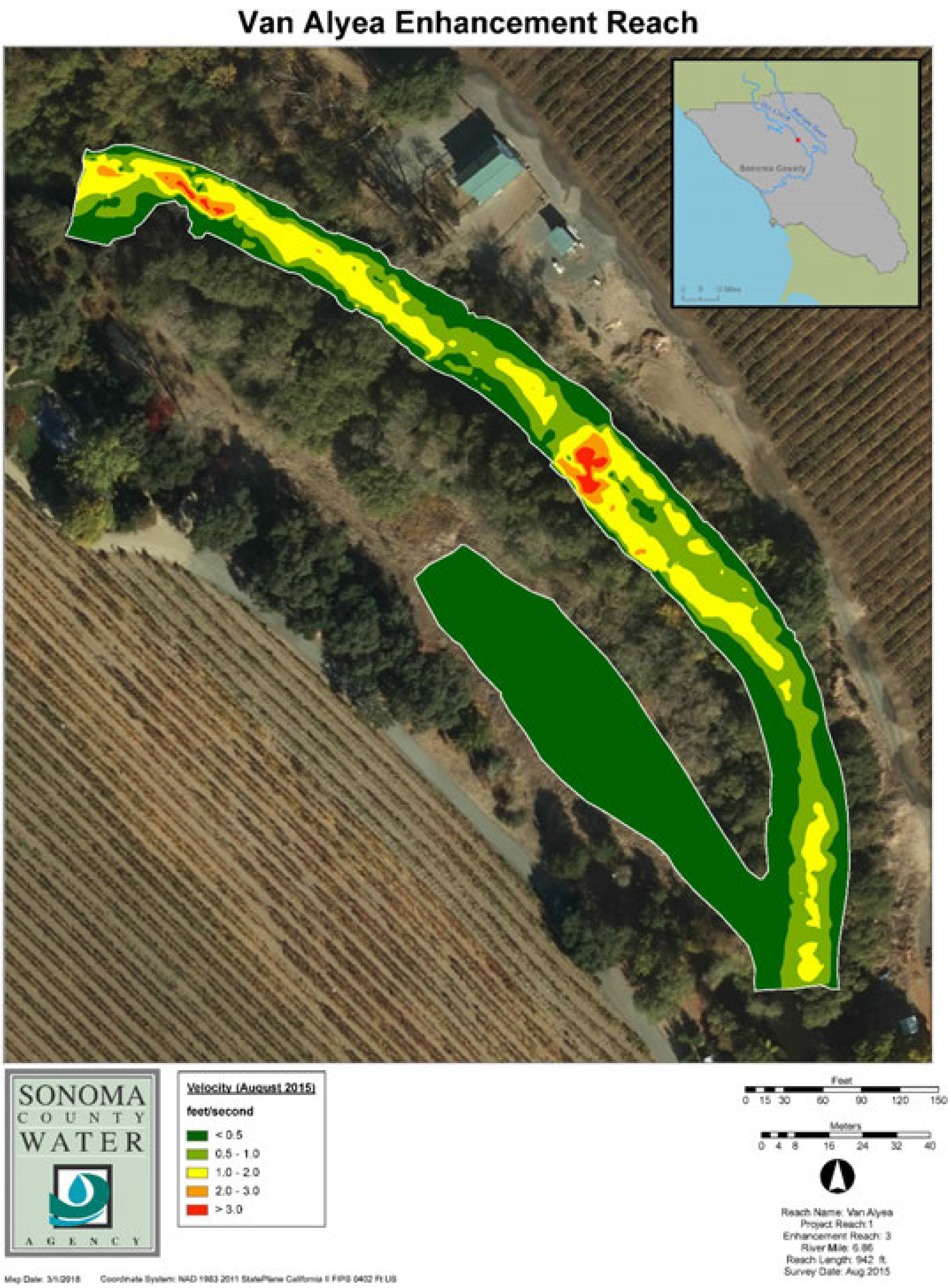

Figure 5.2.9. Measured water velocity within the Van Alyea habitat enhancement reach during August 2015.

Figure 5.2.10. Area of optimal water velocity within the Van Alyea habitat enhancement reach during August 2015.

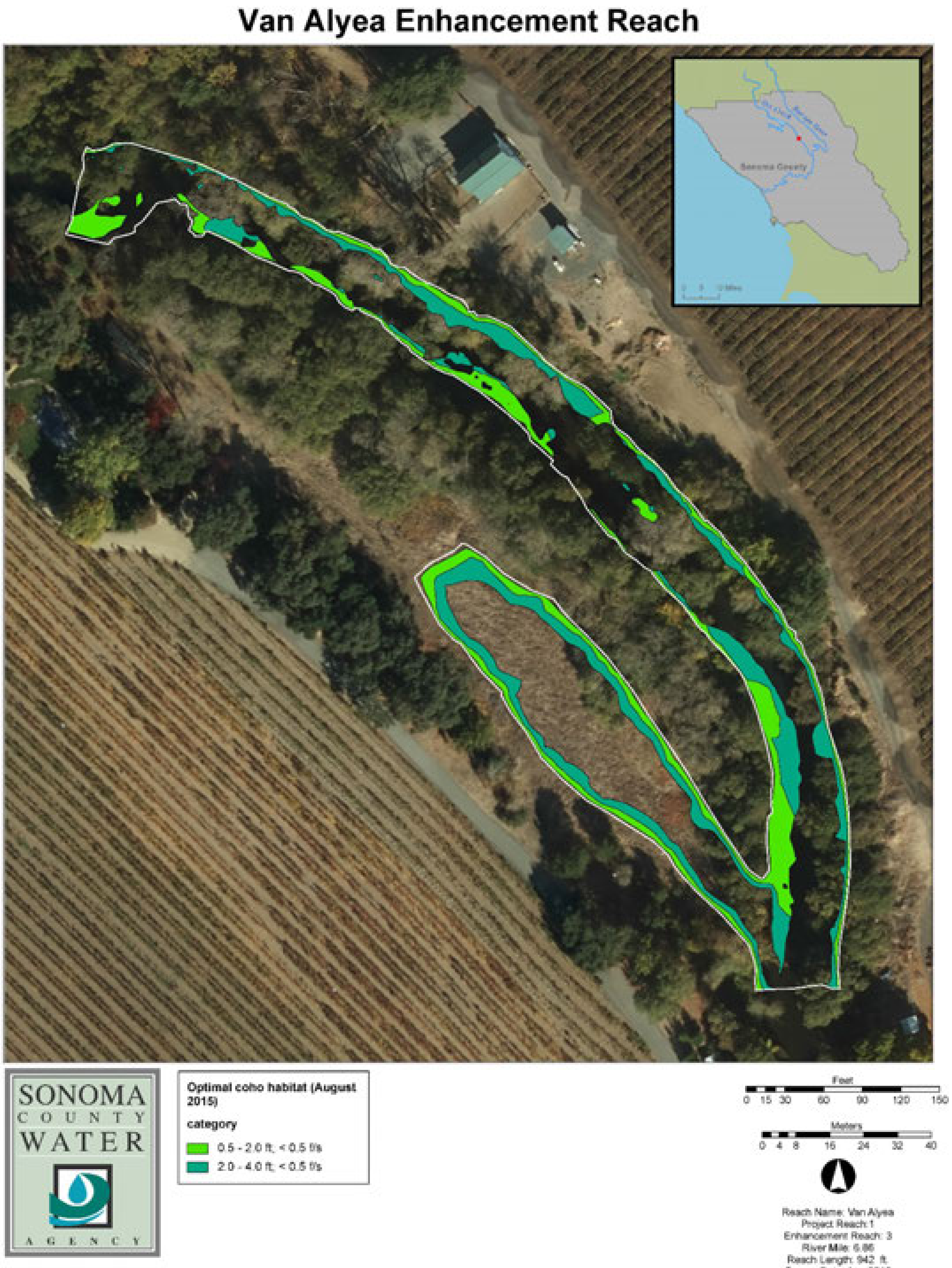

Figure 5.2.11. Area and location of optimal fry (<0.5 f/s, 0.5-2.0 ft) and parr (<0.5 f/s, 2.0-4.0 ft) habitat within the Van Alyea habitat enhancement reach during August 2015.

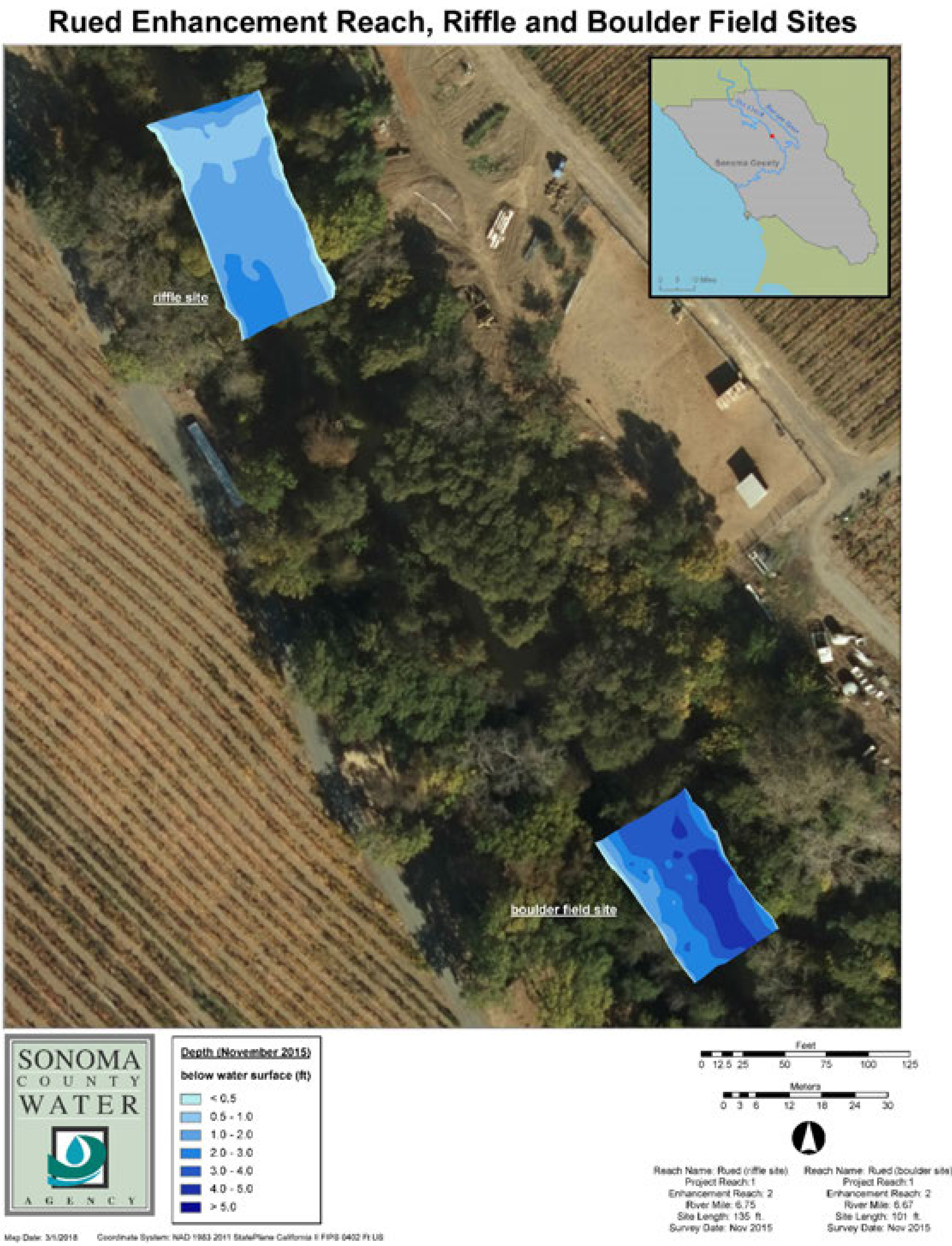

Figure 5.2.12. Measured water depth within the Rued habitat enhancement reach during November 2015.

Figure 5.2.13. Area of optimal water depth within the Rued habitat enhancement reach during November 2015.

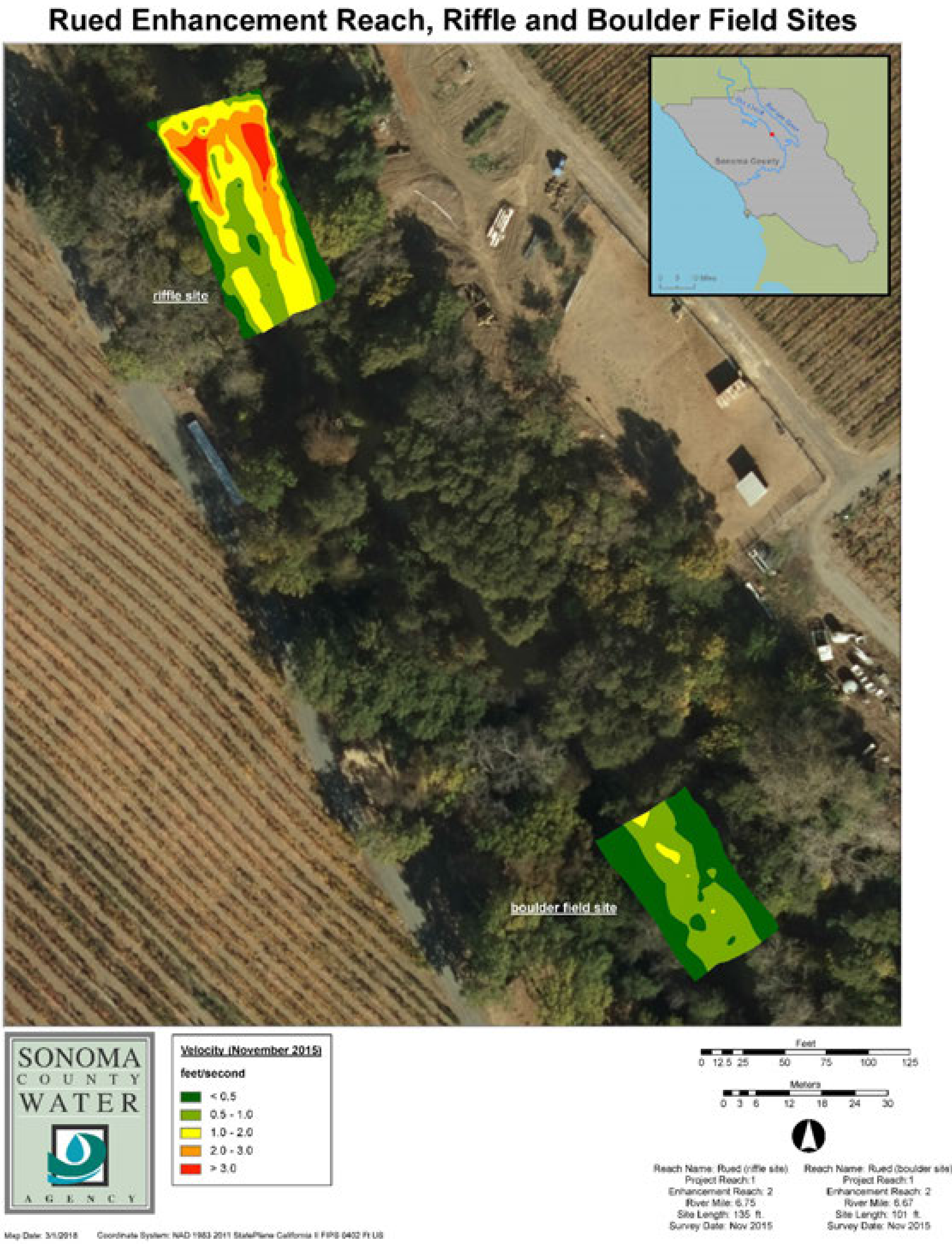

Figure 5.2.14. Measured water velocity within the Rued habitat enhancement reach during November 2015.

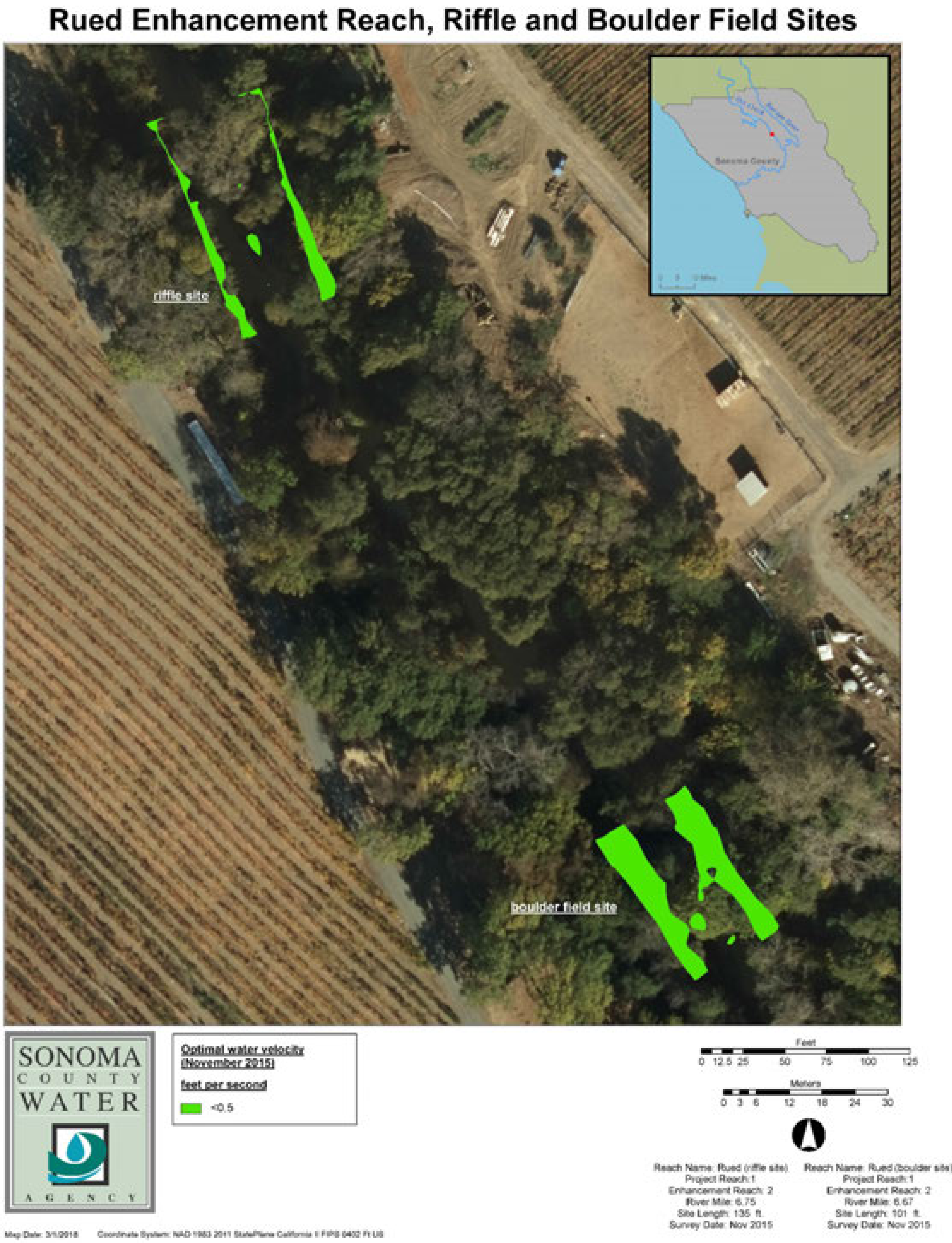

Figure 5.2.15. Area of optimal water velocity within the Rued habitat enhancement reach during November 2015.

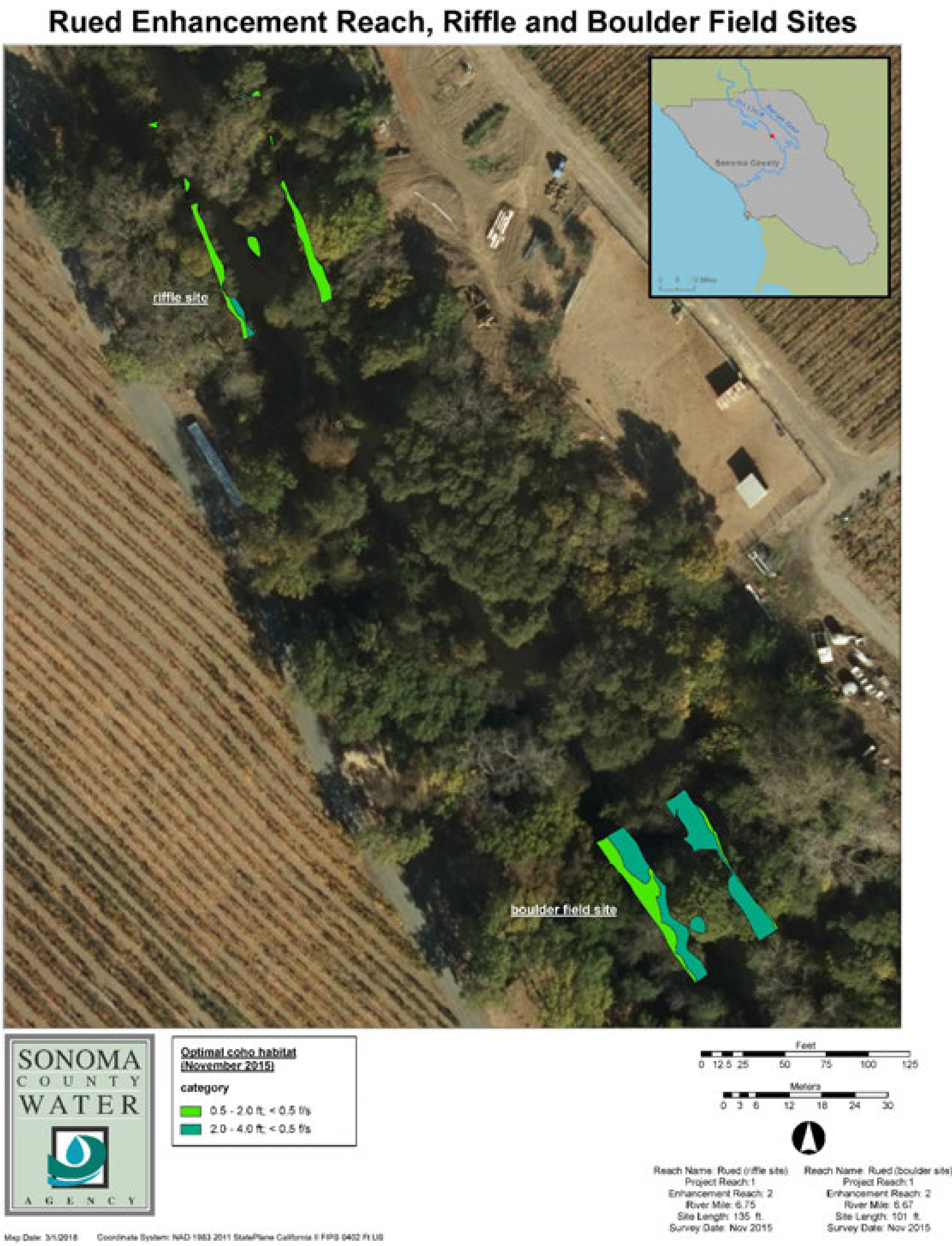

Figure 5.2.16. Area and location of optimal fry (<0.5 f/s, 0.5-2.0 ft) and parr (<0.5 f/s, 2.0-4.0 ft) habitat within the Rued habitat enhancement reach during November 2015.

Figure 5.2.17. Measured water depth within the Army Corps of Engineers habitat enhancement reach during November 2015.

Figure 5.2.18. Area of optimal water depth within the Army Corps of Engineers habitat enhancement reach during November 2015.

Figure 5.2.19. Measured water velocity within the Army Corps of Engineers habitat enhancement reach during November 2015.

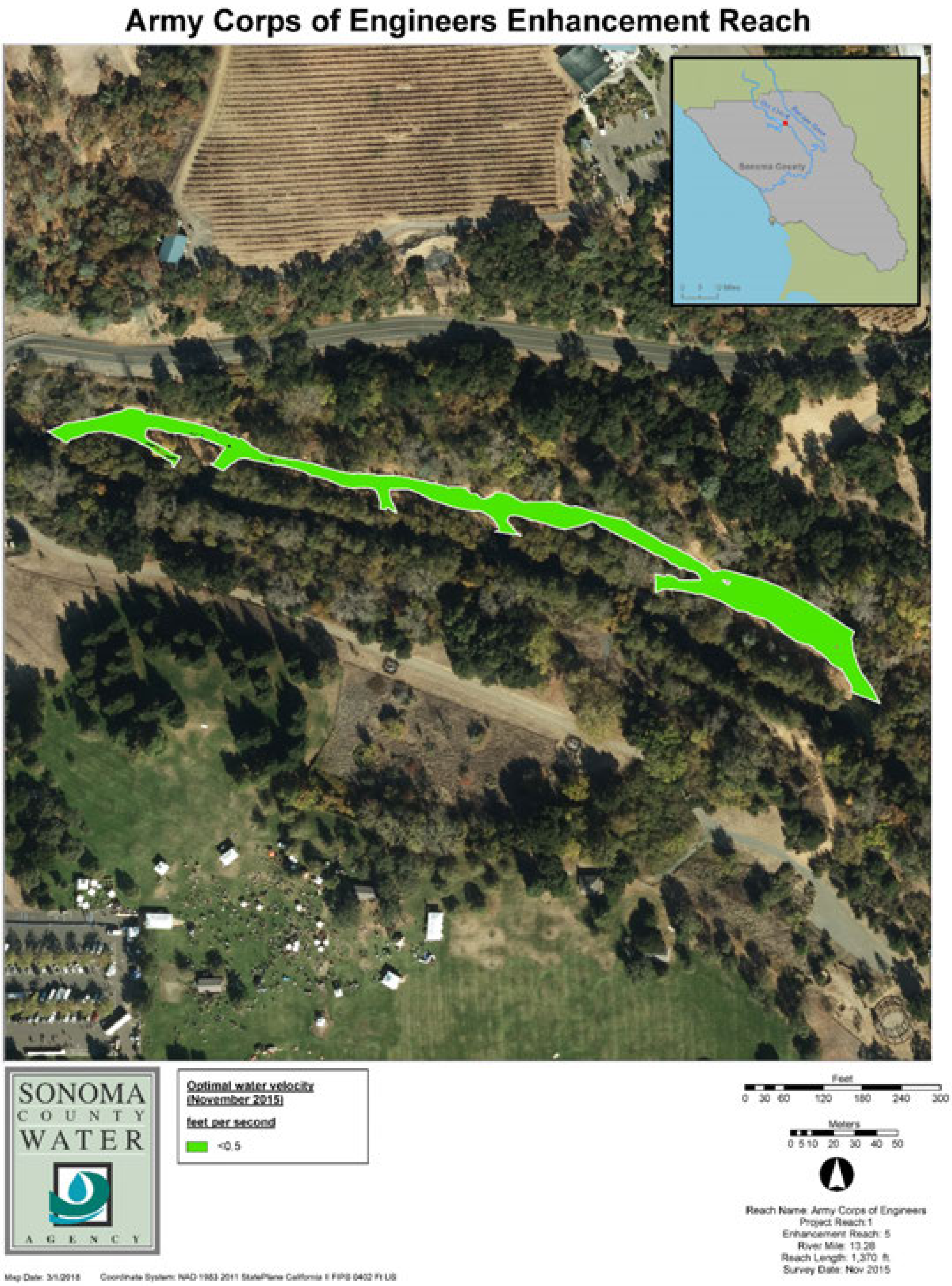

Figure 5.2.20. Area of optimal water velocity within the Army Corps of Engineers habitat enhancement reach during November 2015.

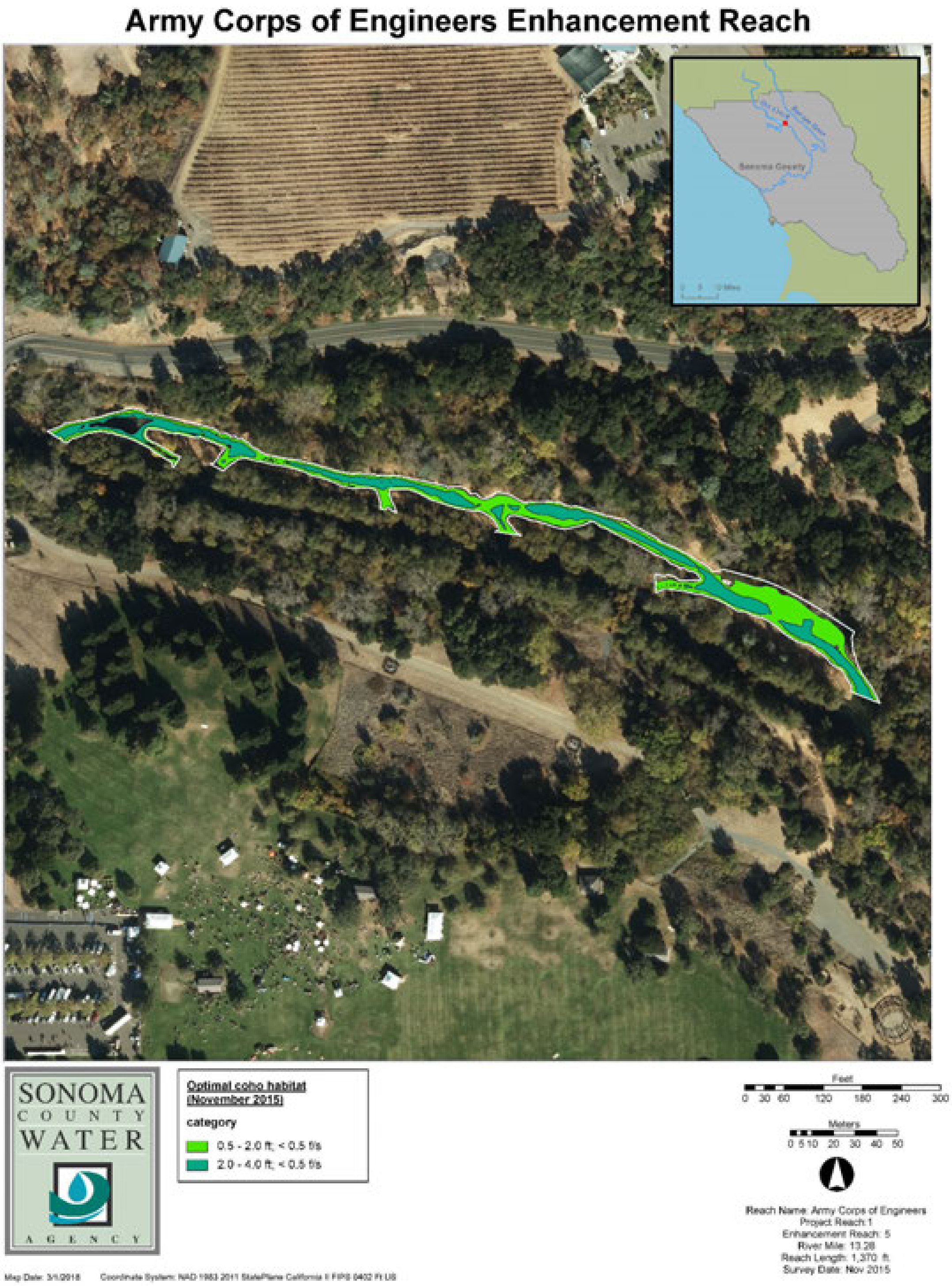

Figure 5.2.21. Area and location of optimal fry (<0.5 f/s, 0.5-2.0 ft) and parr (<0.5 f/s, 2.0-4.0 ft) habitat within the Army Corps of Engineers habitat enhancement reach during November 2015.

Shelter Value

Field Crews inventoried instream habitat units in the four enhancement reaches of the Demonstration Mile in August and November 2015 (Table 5.2.3-Table 5.2.6, Figure 5.2.22-Figure 5.2.27). The crews determined habitat type, and assigned shelter scores and estimated percent overhead cover to determine a shelter value for each habitat unit. All habitat units observed within the Army Corps of Engineers enhancement reach exceeded a shelter value of 80 (Table 5.2.6, Figure 5.2.28 and Figure 5.2.29) considered optimal within the Dry Creek Adaptive Management Plan (Porter et al. 2014). The Farrow Wallace enhancement reach contained the most habitat units not meeting a shelter value of 80, with several units rating a value of 10. (Table 5.2.3, Figure 5.2.22 and Figure 5.2.23). The Van Alyea enhancement reach also contained several habitat units not meeting a shelter value of 80 (Table 5.2.4, Figure 5.2.24 and Figure 5.2.25). All habitat units with shelter values less than 80 occur within mainstem Dry Creek and lack adequate instream cover and overhead cover. Off-channel areas constructed as part of the Dry Creek Habitat Enhancement Project, such as backwaters within the Farrow Wallace and Van Alyea enhancement reaches and the side channel constructed within the Army Corps of Engineers enhancement reach, showed the highest shelter values, due to complex instream cover and abundant overhead cover. The Rued riffle and boulder sites occurred entirely within the mainstem, but still supported shelter values greater than 80 (Table 5.2.5, Figure 5.2.26 and Figure 5.2.27).

Table 5.2.3. Habitat, types, shelter score, percent cover, and shelter value for habitat units within the Farrow Wallace enhancement reach in August 2015.

Habitat Unit #	Habitat Type	Shelter Score	Percent Cover	Shelter Value
U1	Flatwater	2	20	40
U2	Pool	3	35	105
U3	Flatwater	3	40	120
U4	Flatwater	3	30	90
U5	Pool	2	25	50
U6	Riffle	2	5	10
U7	Flatwater	2	5	10
U8	Flatwater	2	5	10
U9	Riffle	2	40	80
U10	Pool	3	25	75
U11	Flatwater	3	30	90
U12	Pool	2	40	80
U13	Flatwater	2	5	10
U14	Backwater	3	80	240
U15	Backwater	3	90	270
U16	Backwater	3	50	150
U17	Riffle	3	60	180
U18	Riffle	3	50	150
U19	Backwater	3	80	240

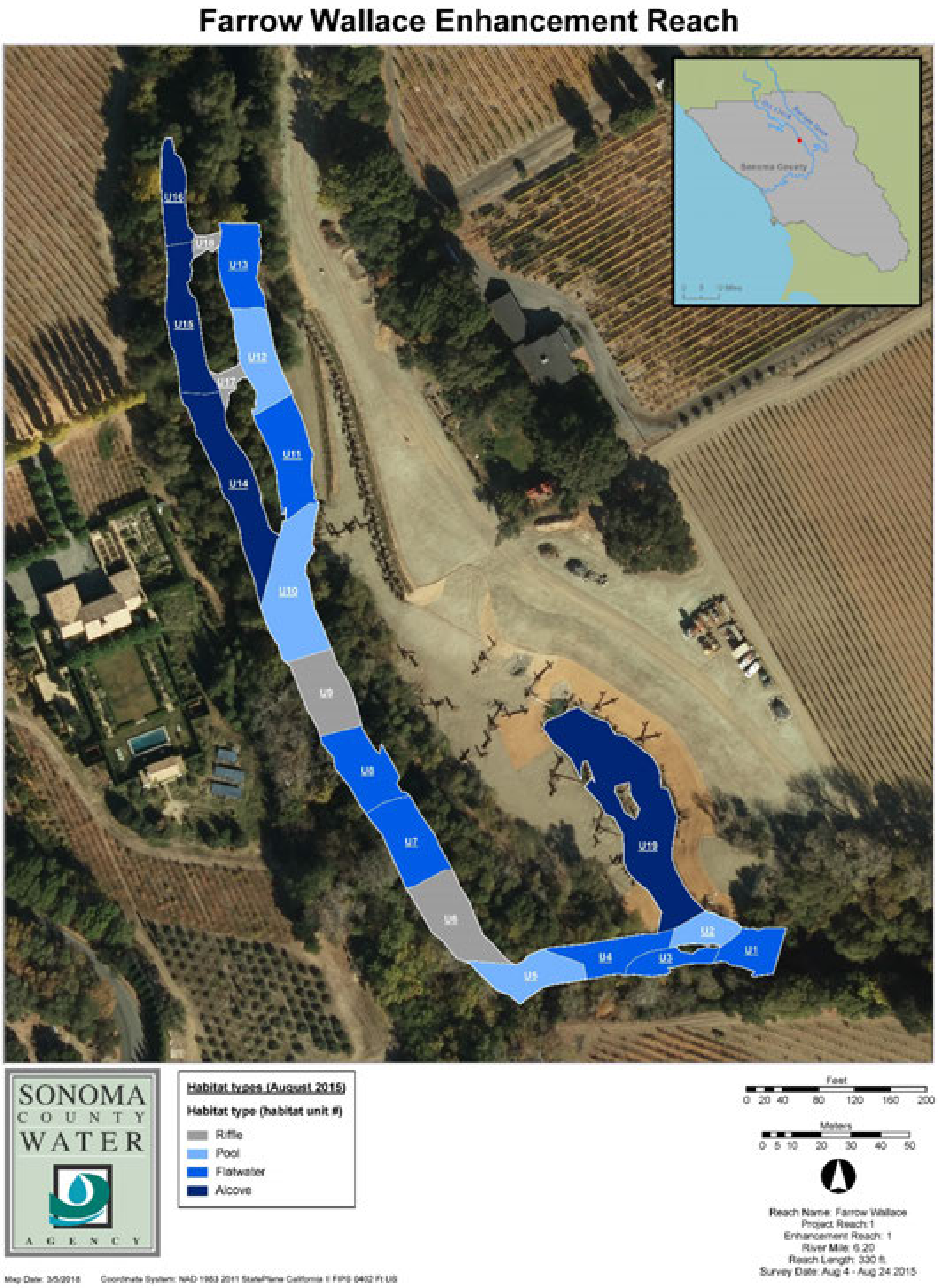

Figure 5.2.22. Habitat unit number and type within the Farrow Wallace enhancement reach in August 2015.

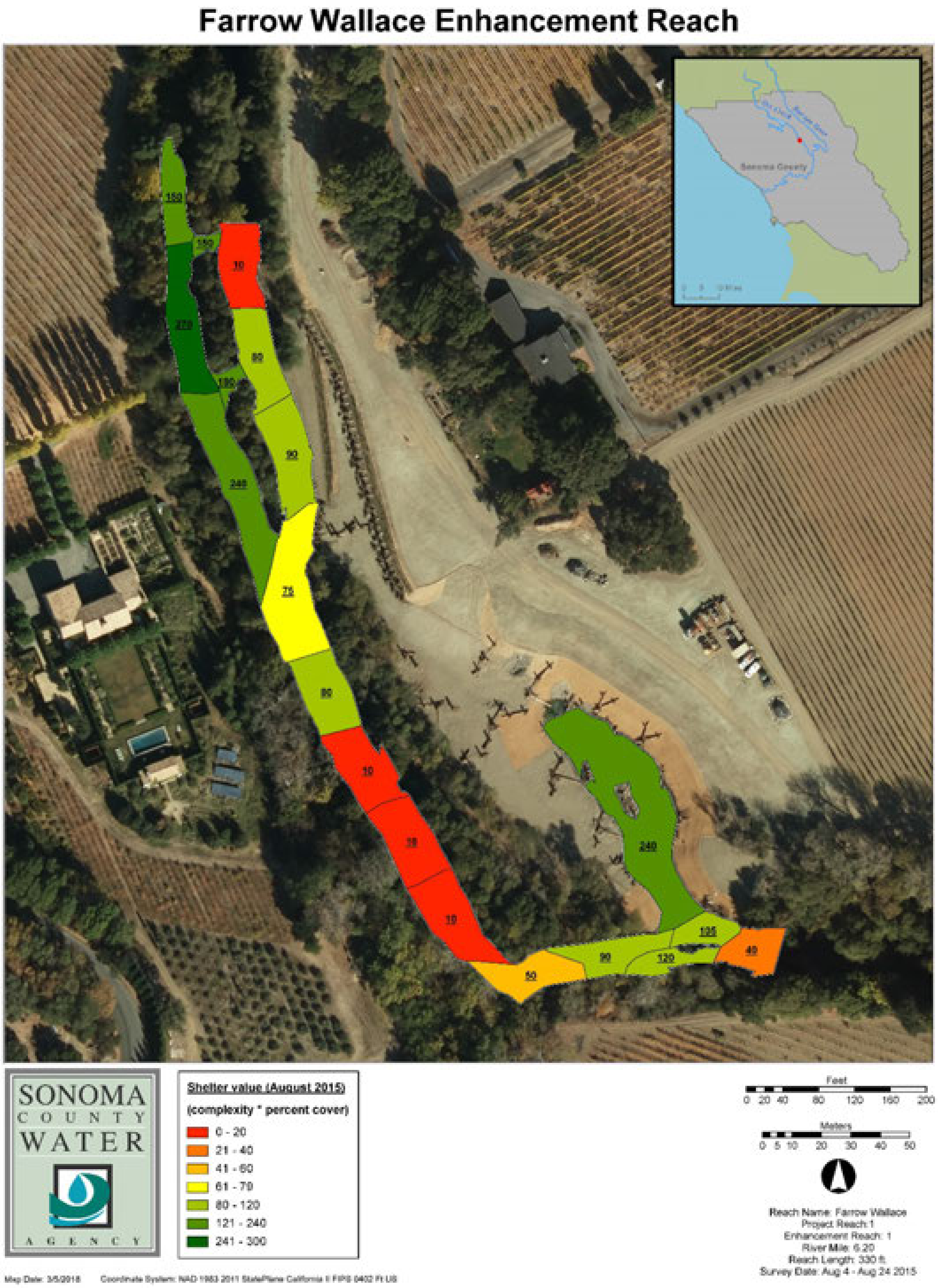

Figure 5.2.23. Habitat unit shelter values for the Farrow Wallace habitat enhancement reach in August 2015.

Table 5.2.4. Habitat, types, shelter score, percent cover, and shelter value for habitat units within the Van Alyea enhancement reach in August 2015.

Habitat Unit #	Habitat Type	Shelter Score	Percent Cover	Shelter Value
U1	Backwater	3	90	270
U2	Flatwater	3	30	90
U3	Flatwater	3	45	135
U4	Flatwater	3	35	105
U5	Flatwater	3	25	75
U6	Riffle	3	30	90
U7	Pool	2	35	70
U8	Pool	3	15	45
U9	Flatwater	1	5	5
U10	Pool	3	20	60
U11	Flatwater	3	40	120
U12	Flatwater	2	25	50

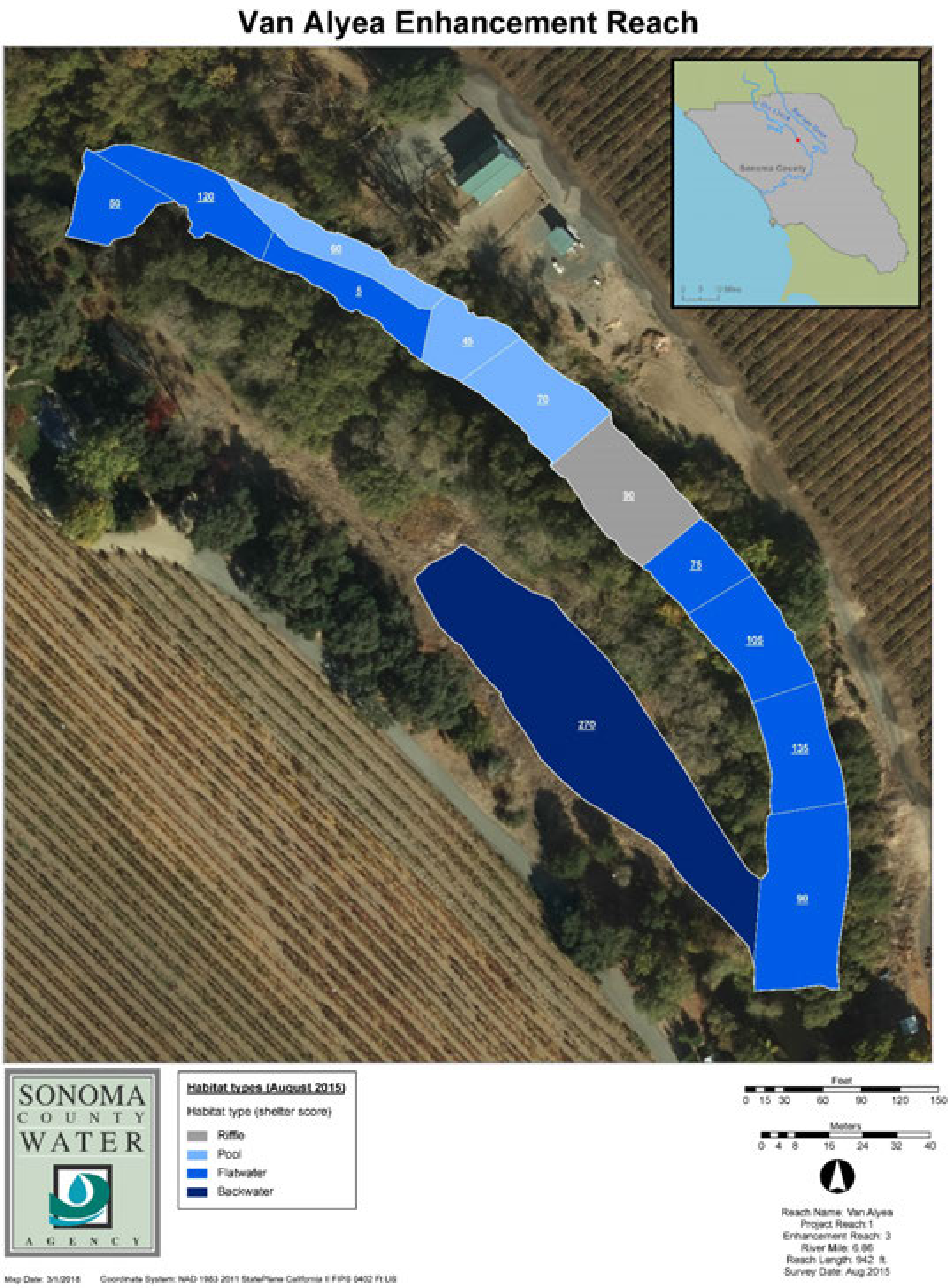

Figure 5.2.24. Habitat unit number and type within the Van Alyea enhancement reach in August 2015.

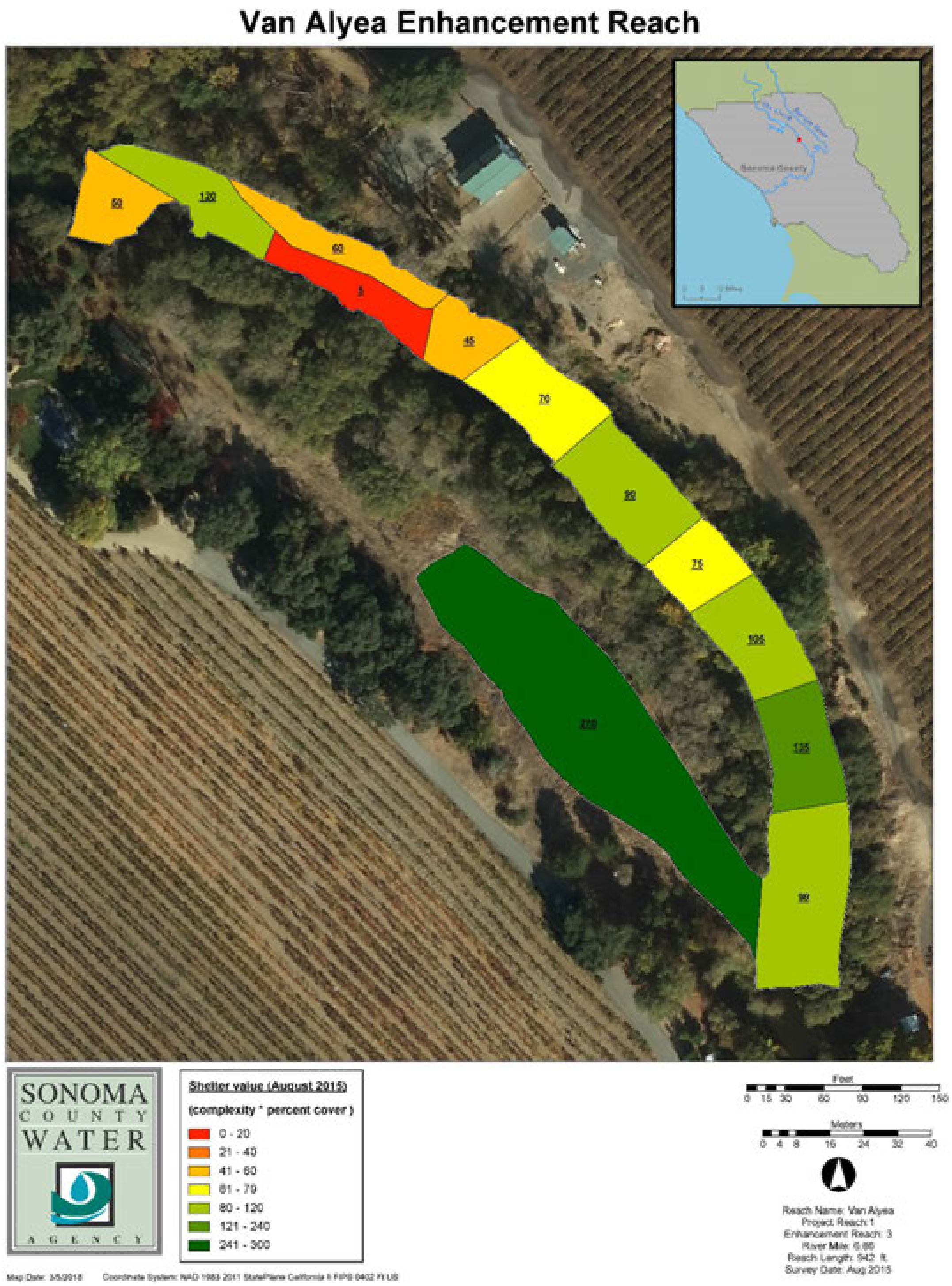

Figure 5.2.25. Habitat unit shelter values for the Van Alyea habitat enhancement reach in August 2015.

Table 5.2.5. Habitat, types, shelter score, percent cover, and shelter value for habitat units within the Rued riffle and boulder enhancement sites in November 2015.

Habitat Unit #	Habitat Type	Shelter Score	Percent Cover	Shelter Value
U1 (riffle site)	Riffle	3	40	120
U1 (boulder site)	Flatwater	3	35	105

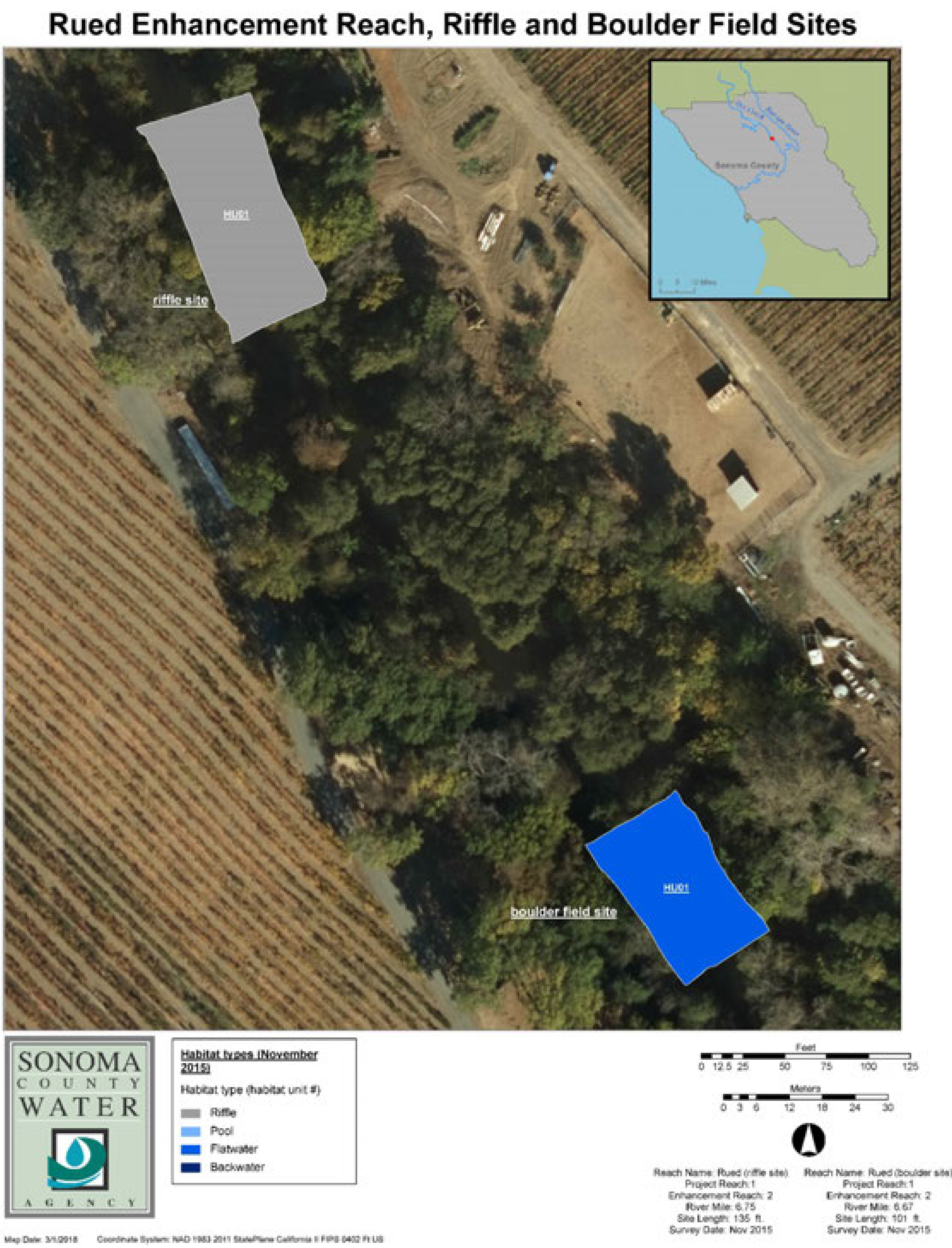

Figure 5.2.26.Habitat unit number and type within the Rued riffle and boulder enhancement sites in November 2015.

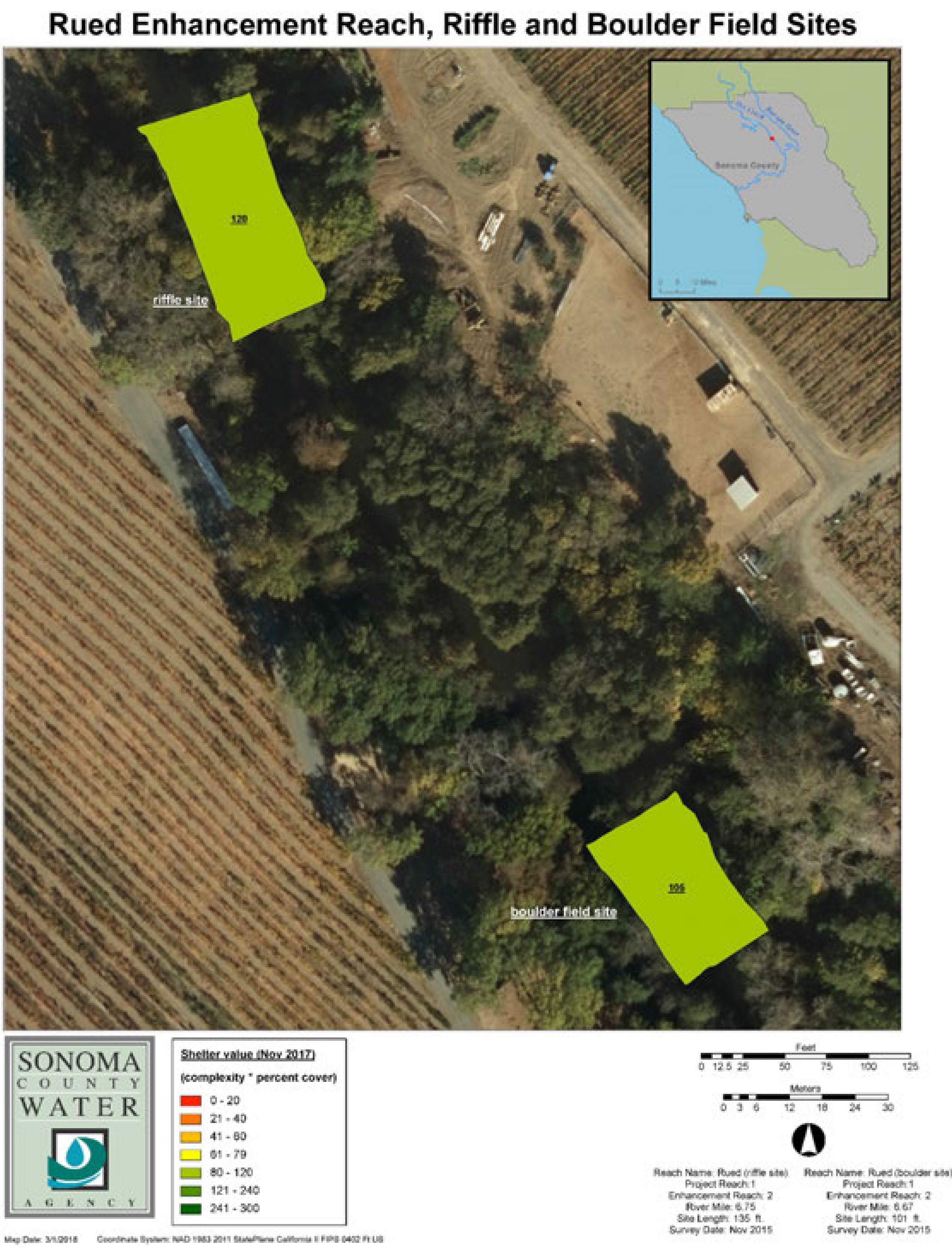

Figure 5.2.27.Habitat unit shelter values for the Rued riffle and boulder enhancement sites in November 2015.

Table 5.2.6. Habitat, types, shelter score, percent cover, and shelter value for habitat units within the Army Corps of Engineers enhancement reach in November 2015.

Habitat Unit #	Habitat Type	Shelter Score	Percent Cover	Shelter Value
U1	Flatwater	3	90	270
U2	Flatwater	3	80	240
U3	Flatwater	3	85	255
U4	Pool	3	95	285
U5	Flatwater	3	95	285
U6	Flatwater	3	80	240
U7	Flatwater	3	75	225
U8	Flatwater	3	85	255
U9	Pool	3	80	240

Figure 5.2.28. Habitat unit number and type within the Army Corps of Engineers enhancement reach in November 2015.

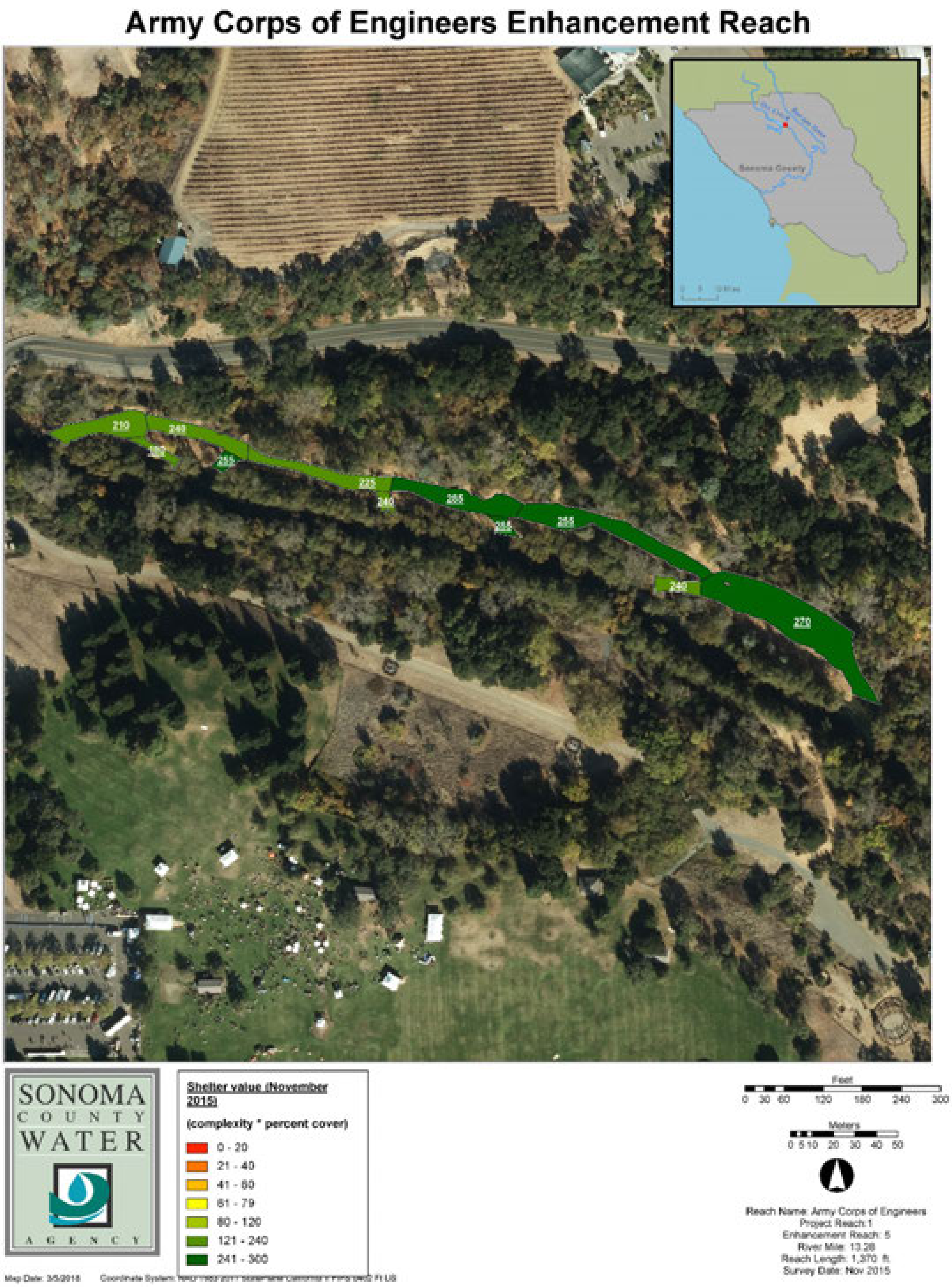

Figure 5.2.29. Habitat unit shelter values for the Army Corps of Engineers habitat enhancement reach in November 2015.

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Porter, M. D., D. M. Marmorek, D. Pickard, and K. Wieckowski. 2014. Dry Creek Adaptive Management Plan (AMP), Version 0.93. Final document prepared by ESSA Technologies Ltd., Vancouver, BC for Sonoma County Water Agency, Santa Rosa CA. 33 pp. + appendices.

5.3 Validation Monitoring

Part of the Adaptive Management Plan (AMP) for validating the effectiveness of habitat enhancement in mainstem Dry Creek calls for a multiscale monitoring approach in both space and time (Porter et al. 2013). The current section of this report focuses on the results of validation monitoring for juvenile and smolt salmonid populations in mainstem Dry Creek in 2015. These data are part of an ongoing pre-construction (baseline) monitoring effort begun in 2008 and outlined in the Reasonable and Prudent Alternative section of NMFS' Russian River Biological Opinion. Construction of the first mile of habitat enhancements in mainstem Dry Creek (the "demonstration project") was completed allowing us to resume sampling efforts in the stream sections affected by construction prior years. Validation monitoring data collected in newly-constructed habitats are reported as well as continued efforts to monitor trends in juvenile and smolt abundance at the reach and watershed scale.

In the Russian River Biological Opinion status and data report year 2009-10 (Manning and Martini-Lamb 2011), the Water Agency outlined six possible metrics that could be considered for validation monitoring of juvenile salmonids with respect to eventual habitat enhancements in the mainstem of Dry Creek: habitat use, abundance (density), size, survival, growth and fidelity (Table 5.3.1). In 2009-2010, a major focus of validation monitoring in Dry Creek was on evaluating the feasibility of sampling methods to accurately estimate each of those metrics while simultaneously attempting to understand how limitations in sampling approaches may affect our ability to validate project success. These same validation metrics and associated limitations and uncertainties have been discussed in the context of the results of those evaluations and are incorporated into the Dry Creek AMP (Porter et al. 2013). The methods currently employed for validation monitoring in Dry Creek are largely based on the outcome of that work (Manning and Martini-Lamb 2011; Martini-Lamb and Manning 2011).

Table 5.3.1. Proposed target life stages, validation metrics, spatio-temporal scale and monitoring tools for validation monitoring in mainstem Dry Creek.

Spatial scale	Target life stage	Target metric(s)	Temporal scale	Primary monitoring tools
Site/feature	Juvenile (non-smolt)	Habitat use, abundance (density), size, growth	Post-construction	Snorkeling, electrofishing, PIT tags and antennas
Reach	Juvenile (non-smolt)	Abundance (density), size, survival, growth, fidelity	Pre-construction (baseline) vs. post-construction	Electrofishing, PIT tags and antennas
Mainstem Dry Creek	Smolt	Abundance	Ongoing to capture long-term trend	Downstream migrant trap, PIT antennas

Methods

In order to address use of newly created habitat by juvenile salmonids at the site (feature) scale, sampling consisted of PIT-tagging in the summer, operation of stationary PIT antennas in the winter and snorkeling in late spring, summer, and early fall. We also conducted mark-recapture electrofishing in enhancement areas to estimate juvenile population density. To better identify how data collected at the site-scale indicate the effect of habitat enhancement we also conducted backpack electrofishing in stream sections (reach-scale) that were not enhanced. Finally, we continued to operate a downstream migrant trap in lower Dry Creek to assess trends in smolt production over time. Broad-scale efforts that are part of the Coastal Monitoring Program (CMP) now being implemented in the Russian River provide a framework for placing our results in the context of watershed-scale patterns in those population metrics identified in Fish Bulletin 180 (Adams et al. 2011).

Habitat utilization

Summer / Fall

We conducted four snorkel surveys in Dry Creek habitat enhancement sites from June to October 2015. Surveys were conducted with two snorkelers working in tandem. From June through September we operated a continuously-recording temperature and dissolved oxygen logger near the mouth of the Farrow backwater (~river km 10.0). During site visits we measured water temperature and dissolved oxygen at 0.5 m depth increments throughout the water column at the location of the continuous loggers allowing us to construct vertical temperature and dissolved oxygen profiles.

Winter

Similar to 2013 and 2014, we operated PIT antennas in newly constructed backwaters during the winter. We installed one antenna each at the downstream end (mouth) of the Farrow (river km ~10.0), Wallace (river km ~10.3) and Van Alyea (river km 11.1) backwaters. In the Wallace backwater, we also operated an antenna mid-way up the backwater. Although antennas did not span the width of the backwaters, they did cover the majority of the wetted width.

The source of PIT-tagged fish included: (1) PIT-tagged juvenile coho from Warm Springs hatchery that were released directly into the backwater in the fall as age-0+ (Table 5.3.2); (2) PIT-tagged juvenile coho from Warm Springs hatchery that were released at other locations throughout the Dry Creek system; (3) wild (natural-origin) juvenile steelhead that were PIT-tagged during other surveys (primarily summer electrofishing surveys). The residence time of PIT-tagged juvenile coho released into the backwater was calculated as the number of days between release date and their final detection date on the PIT antenna. We also detected some of these fish at stationary PIT antenna locations downstream of the backwaters.

Table 5.3.2. Number of coho young-of-the-year released from Warm Springs Hatchery in or near the off-channel habitats constructed on Dry Creek, 2013 and 2014. Coho young-of-the-year were not released near the off-channel habitats in 2015

Mainstem or Off-channel	Release Site	Release River Km	2013	2014
Mainstem	Adjacent to Farrow backwater	9.90		200
Off-channel	Farrow backwater	10.00	759	632
Off-channel	Wallace backwater	10.34		277
Mainstem	Adjacent to Wallace backwater	10.70		200
Mainstem	Adjacent to Quivira backwater	11.62		825
Off-channel	Reach 15 side channel	21.45	250	635

Late summer population density

Site-scale sampling

During late August through early October, we sampled 5 sites within habitat enhancement areas by making a backpack electrofishing pass on day 1 (the marking event) and a second pass through the site two days later (the recapture event). Individuals captured on day 1 were PIT-tagged, released near their capture location and subject to recapture on day 2. From these paired sampling events, we used the Petersen mark-recapture model in Program MARK (White and Burnham 1999) to estimate end of summer abundance ($\widehat{N}$) in 5 sites within habitat enhanced areas. Provided recapture probability, mortality and the proportion of fish leaving the section between the marking and recapture events is the same for the marked group as it is for the unmarked group, the abundance estimates from the paired mark and recapture events in early autumn will be unbiased (White et al. 1982). Density estimates were calculated as the quotient of $\widehat{N}$ and wetted area of the site.

Reach-scale sampling

The Biological Opinion as well as the primary literature (e.g., Roni 2005) acknowledge the problem of biological monitoring that is too limited in time or space to accurately detect changes in population that may result from artificial habitat enhancements. For this reason, we added to our sampling efforts in 2015 by taking advantage of the spatially balanced random sampling framework afforded by the generalized random tessellation stratified (GRTS) framework outlined for the California Coastal Salmonid Monitoring Program (CMP, Adams et al. 2011) that is now being implemented in the Russian River. Sampling reaches in this manner over time will allow us to place our results in a broader spatial context thereby facilitating more accurate validation of the effectiveness of habitat enhancement measures in Dry Creek (Figure 5.3.1). Towards that end, we sampled one stream section in each of nine “GRTS” reaches defined in mainstem Dry Creek for CMP monitoring. We sampled using methods similar to those described for site-scale electrofishing so that we could estimate juvenile steelhead abundance using the Petersen mark-recapture model. Stream sections (sub-reaches) were typically longer (546 to 1050 feet) than sites sampled during site-scale sampling. To allow comparisons with data from prior years and to provide some pre-construction monitoring data, we sampled four additional stream sections (one in the lower reach and three in the middle reach).

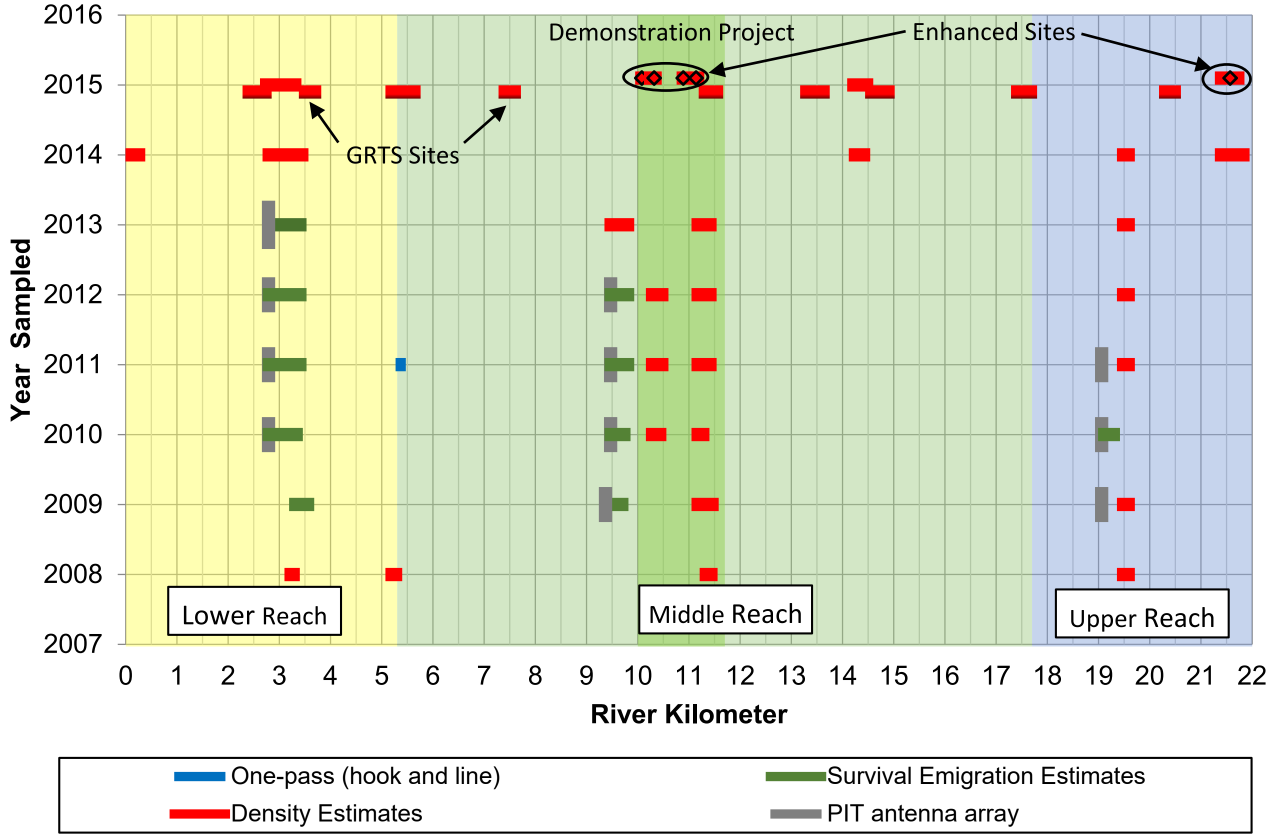


Figure 5.3.1. Years sampled and river kilometer (from the mouth) where juvenile steelhead populations were sampled in mainstem Dry Creek, 2008-2015. Line length for each site is scaled to the length of stream sampled. Data collected at the site scale were analyzed using mark-recapture (either a multiple-pass depletion or Petersen model) and reach scale data collected in 2009 were analyzed with the core-sampling approach (see Manning and Martini-Lamb 2011 for details) while reach scale data collected in 2011-13 were analyzed with the multistate model using program MARK (White and Burnham 1999) to estimate survival and emigration. The darker green-shaded area indicates the stream section that has been targeted to receive the first mile of habitat enhancements (the "Demonstration Project"). We adopted the geomorphically-based reach designations identified by Inter-Fluve (2011) for defining reaches for use in summarizing density estimates.

Smolt abundance

A rotary screw trap with a 1.5 m diameter cone was anchored to the Westside Road bridge, located 3.3 km upstream from the confluence of Dry Creek and the Russian River. Wood-frame mesh panels were installed adjacent to the rotary screw trap in order to divert downstream migrating salmonids into the trap that may have otherwise avoided the trap.

Fish handling methods and protocols were similar to those used in previous years (see Manning and Martini-Lamb 2011). Fish captured in the trap were identified to species and enumerated. A subsample of each species was anesthetized and measured for fork length each day, and a subsample of salmonid species was weighed each week. With the exception of up to 50 Chinook salmon smolts each day, all fish were released downstream of the first riffle located downstream of the trap.

Each day, up to 50 Chinook smolts (≥60 mm) were marked and released upstream of the trap for the purpose of estimating trap efficiency and constructing a population estimate. Fin clips were used Friday through Sunday to mark fish while PIT tags were used Monday through Thursday. PIT-tagged fish provided the potential to evaluate migration mortality and migration time as fish were detected at downstream monitoring sites (i.e., Northwood and Duncans Mills PIT antenna arrays). Fin clipped and PIT-tagged fish that were recaptured in the trap were noted and released downstream (the lengths and weights of recaptured fish were not recorded a second time). The population estimate of Chinook salmon smolts produced in the Dry Creek watershed upstream of the trap were based on recapture rates of fin clipped and PIT tagged fish. The abundance estimate of Chinook smolts reported in 2015 applies to the period of time that fish were marked and released upstream of the trap (March 26-July 31).

Results

Habitat utilization

Summer / Fall

Counts from snorkel surveys of juvenile salmonids in the Dry Creek habitat enhancement areas were low (Table 5.3.3), but, rooted aquatic vegetation and algae growth, which resulted in poor visibility, adversely affected our ability to observe juvenile salmonids (Figure 5.3.2). Evidence of vegetation impacts on snorkeling visibility is clear in light of snorkel and electrofishing surveys conducted in 2014 in the reach 15 side channel. The number of fish observed in the October 2014 during a snorkel survey (34) was far less than the 351 steelhead captured by electrofishing in the same stream section a few days earlier (Martini-Lamb and Manning 2015).

Table 5.3.3. Number of juvenile steelhead observed during snorkel surveys at Dry Creek habitat enhancement sites, 2015 (no coho salmon were observed during summer snorkel surveys).

Site	Date	Steelhead
Farrow backwater	6/8/2015	0
	7/14/2015	1
	9/8/2015	2
	10/5/2015	5
Wallace backwater	6/8/2015	0
	7/14/2015	3
	9/8/2015	3
	10/5/2015	12
Mascherini bank stabilization	6/8/2015	1
Van Alyea Backwater	6/8/2015	0
	7/14/2015	0
	9/8/2015	17
	10/5/2015	21
USACE (reach 15) side channel	6/8/2015	0
	7/14/2015	1

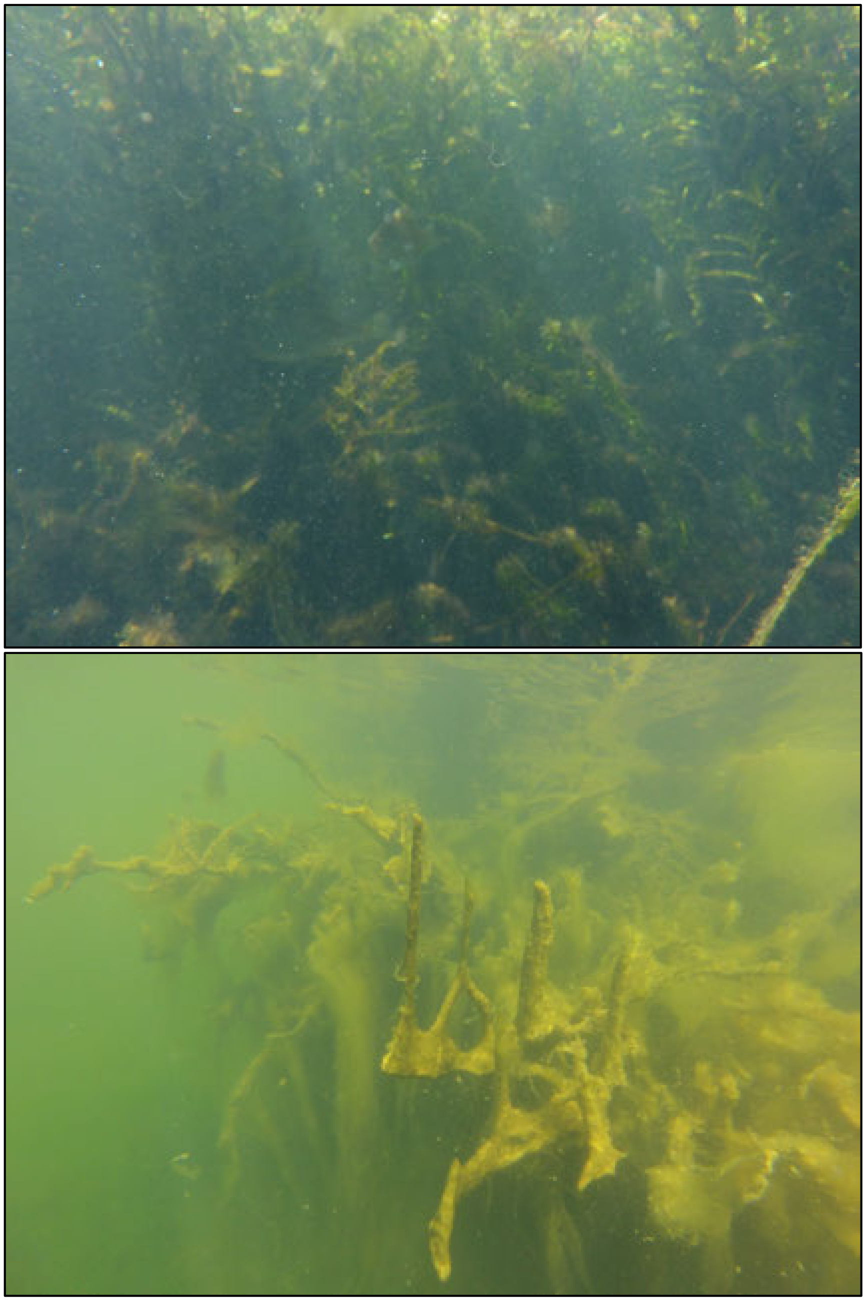


Figure 5.3.2. Underwater photos in Farrow backwater illustrating dense rooted vegetation growth (upper photo) and algae growth (lower photo).

While it is impossible to quantify the effect on juvenile salmonid counts from deteriorating visibility caused by increasing amounts of algae in the Farrow backwater, we suspect that low dissolved oxygen did impact salmonid use. Mean daily dissolved oxygen and vertical water quality profiles showed deteriorating conditions both seasonally and throughout the water column (Figure 5.3.3). However, dissolved oxygen was favorable in the other habitat enhancement sites.

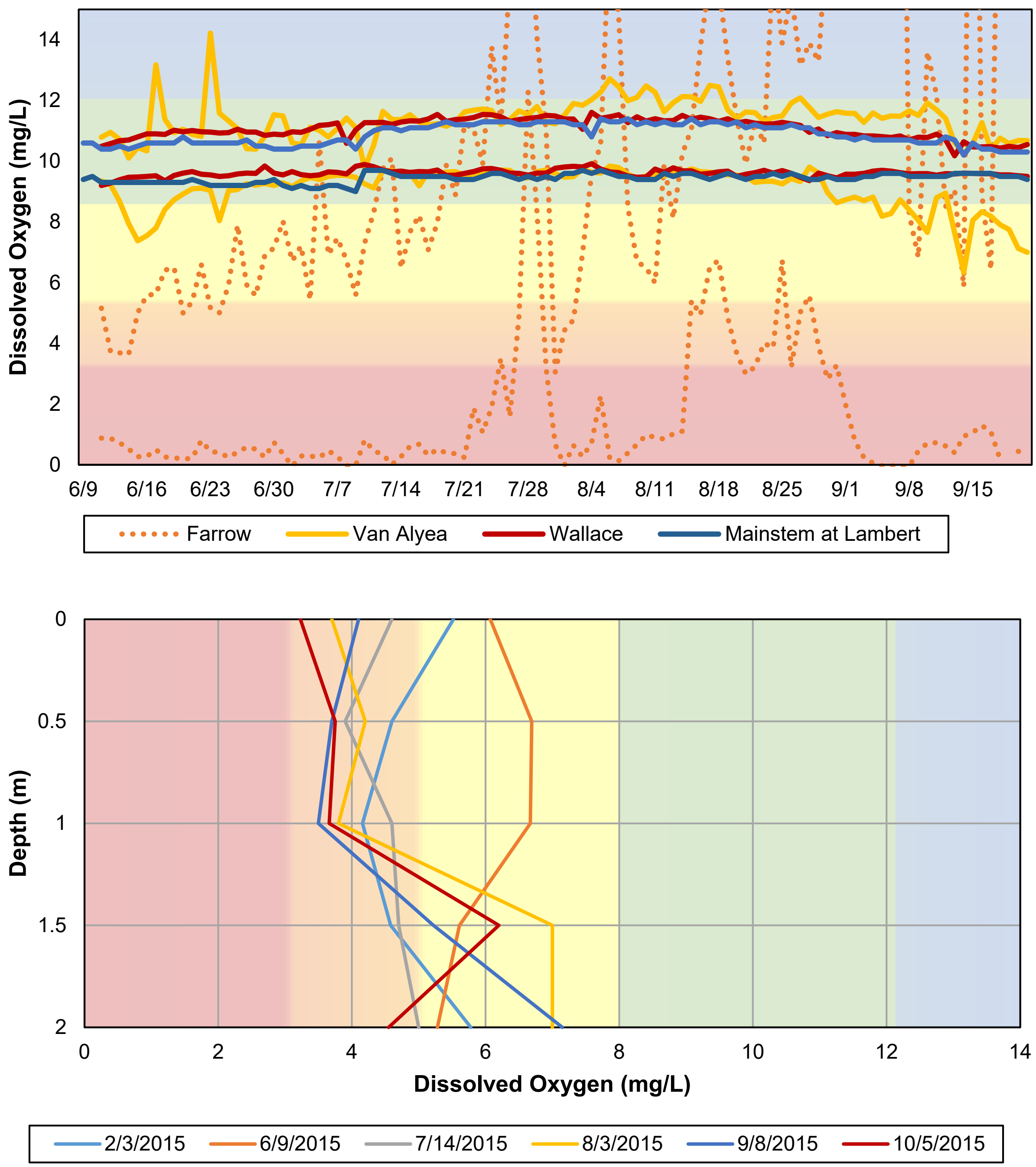


Figure 5.3.3. Daily mean, minimum and maximum dissolved oxygen and water temperature collected with a stationary, continuously recording probe in Farrow, Van Alyea, Wallace, and mainstem Dry Creek (upper panel) and dissolved oxygen from vertical water quality profiles collected with a handheld probe at 0.5 m depth increments at Farrow(lower panel).

Winter

PIT tag antennas installed at the mouth of the habitat enhancements were used to detect PIT tagged coho and steelhead entering habitat enhancement sites. In late 2014, PIT-tagged coho parr were released into Dry Creek habitat enhancement sites. A summary of detections of these fish were reported in Martini-Lamb and Manning (2016). Here we report all detections that occurred in 2015, some of which were from PIT tagged coho released in the 2014 release group.

PIT tagged fish were frequently detected in the Dry Creek Habitat enhancement sites during the winter and fall of 2015 when antennas were operated. In 2015, 203 PIT-tagged individuals were detected a total of 5,078 times in the newly-created backwaters over a wide range of flows (Figure 5.3.4). These PIT-tagged fish likely represent only a small portion of the fish that took advantage of the habitat enhancement sites as most fish occupying Dry Creek whether hatchery or wild are untagged. Of these 203 PIT tagged fish 157 were coho parr tagged and released into Dry Creek as part of the Coho Salmon Conservation Program (CSCP) and 46 were steelhead that were tagged during electrofishing surveys mainstem Dry Creek (Table 5.3.4). Of the 203 individuals detected, at least 30 days elapsed between the first and last detection for 41 fish and more than 100 days for 16 individuals suggesting that a significant number of fish likely took advantage of the habitat enhancements for extended periods of time. Of the tags detected in the habitat enhancement sites, 18%, 27%, and 55% were detected in the Farrow, Van Alyea, and Wallace backwaters, respectively. We interpret this to mean that all three habitat enhancement sites provided juvenile salmonid habitat.

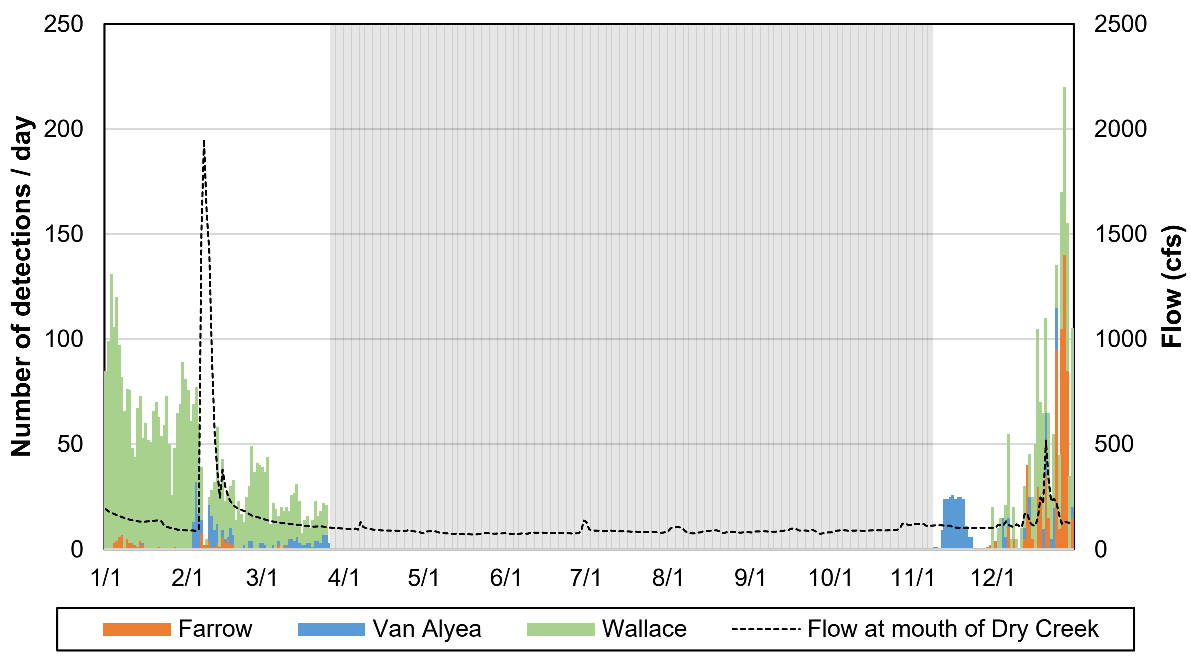


Figure 5.3.4. Number of detections per day of 203 individual juvenile coho and steelhead that entered the Dry Creek Habitat enhancement sites. Also shown is flow at the mouth of Dry Creek (from the USGS gage number 11465350). Dates the PIT tag antennas were not operated are shown in gray.

Table 5.3.4. The number of individual fish detected on PIT tag antennas in the Dry Creek habitat enhancement sites in 2015. The PIT tags for which we do not know the species are likely coho smolts released by the CSCP in the Dry Creek Basin.

Detection Site	coho salmon	steelhead
Farrow backwater PIT	21	12
Van Alyea backwater PIT	50	7
Wallace backwater PIT	86	27
Total	157	46

Data from 2015 also illustrate the importance to salmonids of off-channel features during the winter which allow individuals to express the many life histories present in steelhead populations. A 123 mm steelhead was captured and tagged at in the Russian River estuary and release at Bridgehaven (estuary river km ~3.0) in July, 2015. This individual was detected moving into Dry Creek (mainstem Russian river km ~53.0) in mid-November, detected in the Wallace backwater in late December (Dry Creek river km ~10.3), corresponding with a flow of 500 cfs, and detected entering Grape Creek in early January (Dry Creek river km ~11.7) where it remained until at least mid-March. Although this movement history is for a single fish, it suggests that the off-channel sites constructed in Dry Creek are important not only to fish originating from locations near the habitat enhancement sites, but also to fish originating from more distant locations.

Late summer population density

Site-scale sampling

The average density of juvenile steelhead in enhancement sites was 0.19 fish/m^2 (range 0.05 fish/m^2 to 0.52 fish/m^2, Figure 5.3.5). We captured a total of three wild coho YOY during electrofishing sampling. These fish were found in river km 21.4 in the USACE constructed side channel.

Reach-scale sampling

The average density of juvenile steelhead in GRTS sub-reaches was 0.09 fish/m^2 (range 0.02 fish/m^2 to 0.16 fish/m^2, Figure 5.3.6). When averaged for all sites within a year, densities in 2015 were 0.15 fish/m^2 lower than the seven year average from 2008-2014 (Figure 5.3.7). For the 2015 data, however, a comparison of the average population density for un-enhanced habitat sites (from reach-specific sampling) to average population density for enhanced habitat sites revealed that average density was nearly twice as high in enhanced sites (Figure 5.3.7).

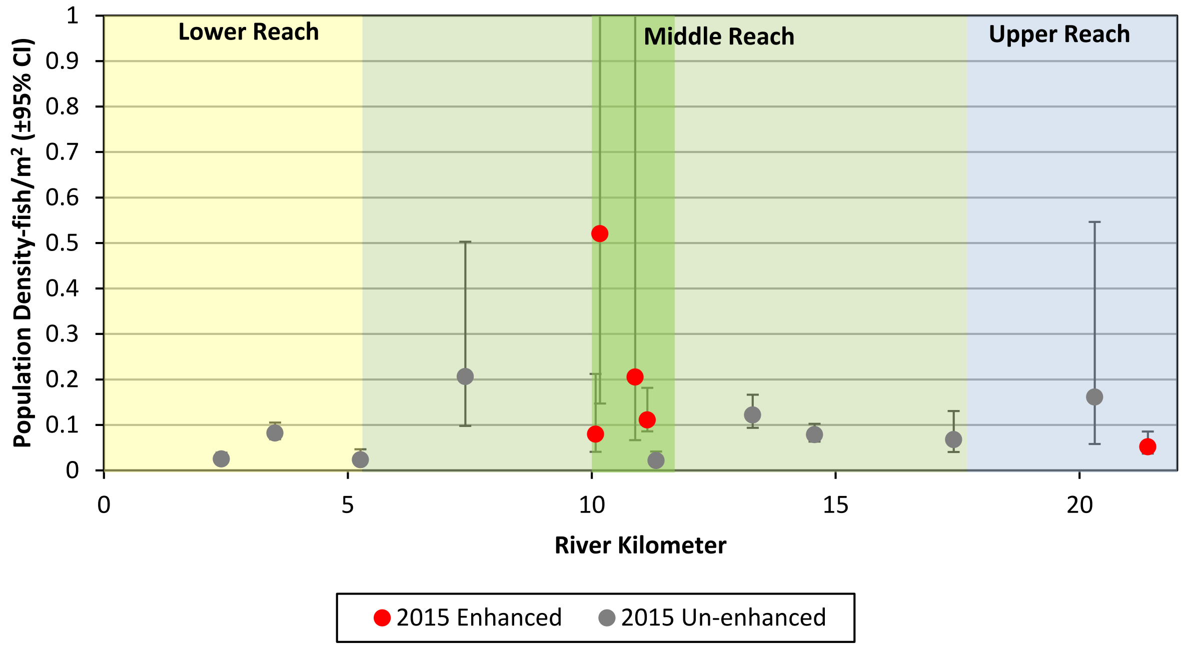


Figure 5.3.5. Estimated density of juvenile steelhead in mainstem Dry Creek, in habitat-enhanced habitat (site-scale monitoring) and un-enhanced habitat (reach-scale monitoring). Estimates are based on the Petersen mark-recapture model.

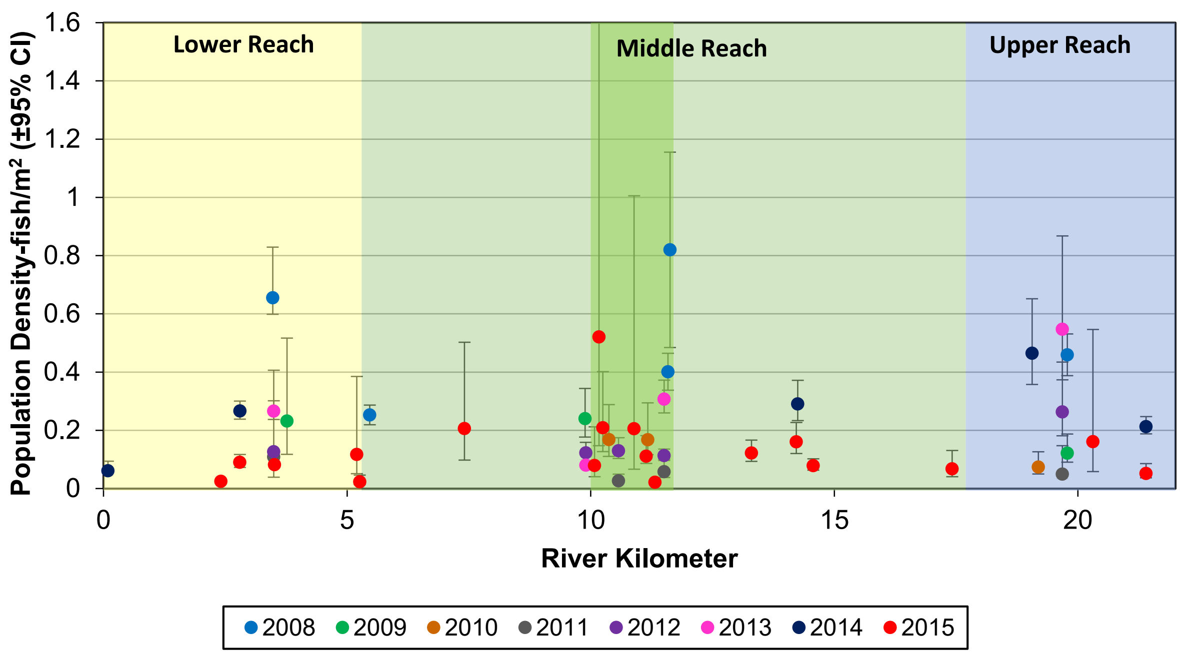


Figure 5.3.6. Estimated density of juvenile steelhead in mainstem Dry Creek, 2008-2015. Estimates are from a variety of approaches all based on mark-recapture models (see text of this and previous Russian River Biological Opinion status and data reports for details).

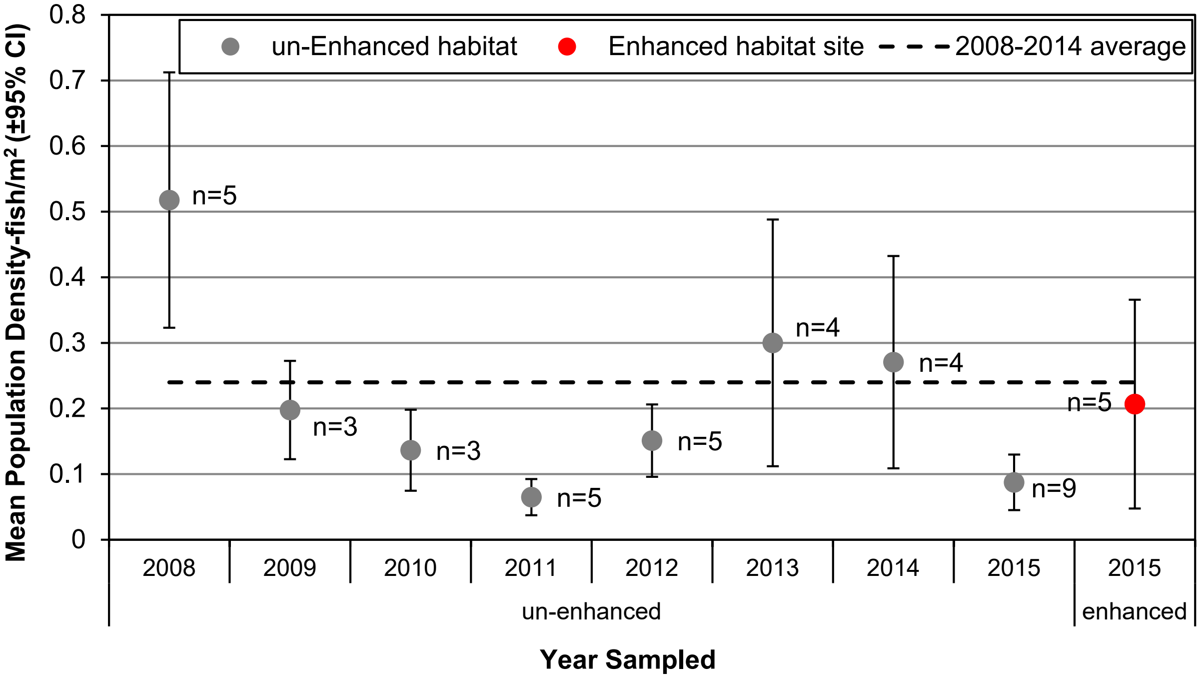


Figure 5.3.7. Mean juvenile steelhead density among all sites sampled within a year in mainstem Dry Creek, 2008-2015. "n" refers to the number of sites sampled per year.

Smolt abundance

We installed the rotary screw trap on March 18 which was the earliest date of operation since we began trap operation in 2009 (Figure 5.3.8). Except for brief periods when trapping was suspended because of high debris loading in the trap from high winds, the trap was checked daily during operation from March until it was removed on July 30.

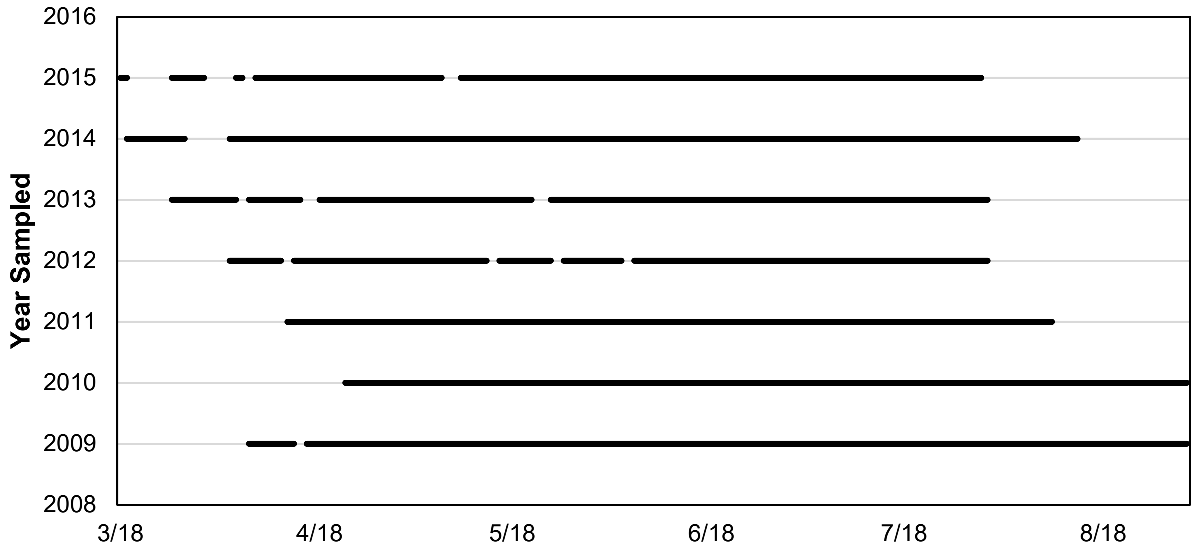


Figure 5.3.8. Begin and end dates and data gaps (spaces in lines) for operation of the Dry Creek downstream migrant trap, 2009-2015.

The peak capture of Chinook smolts (812) occurred during the week of 4/9 (Figure 5.3.9). Based on the estimated average weekly capture efficiency (range: 8% to 34%), the resulting population size of Chinook salmon smolts passing the Dry Creek trap between March 19 and July 30 was 27,053 (±95% CI: 3,619, Figure 5.3.10). This is the smallest population estimate since we began trapping Dry Creek in 2009.

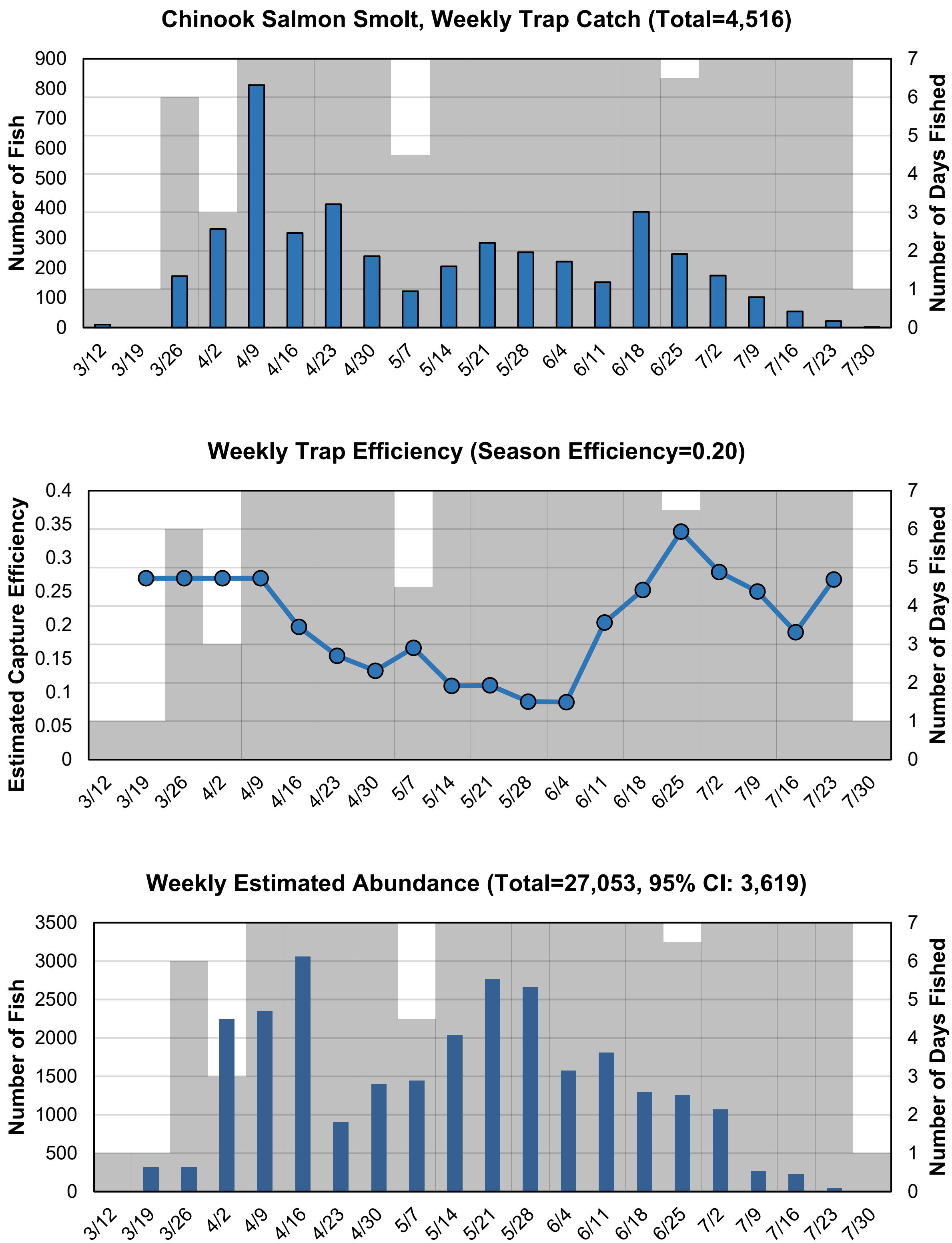


Figure 5.3.9. Weekly trap catch (upper panel), estimated average weekly capture efficiency (middle panel) and population estimate of Chinook salmon smolts in the Dry Creek rotary screw trap (lower panel), 2015. Estimates are from DARR (Bjorkstedt 2005). The number of days each week the trap was fished is represented by the shaded area.

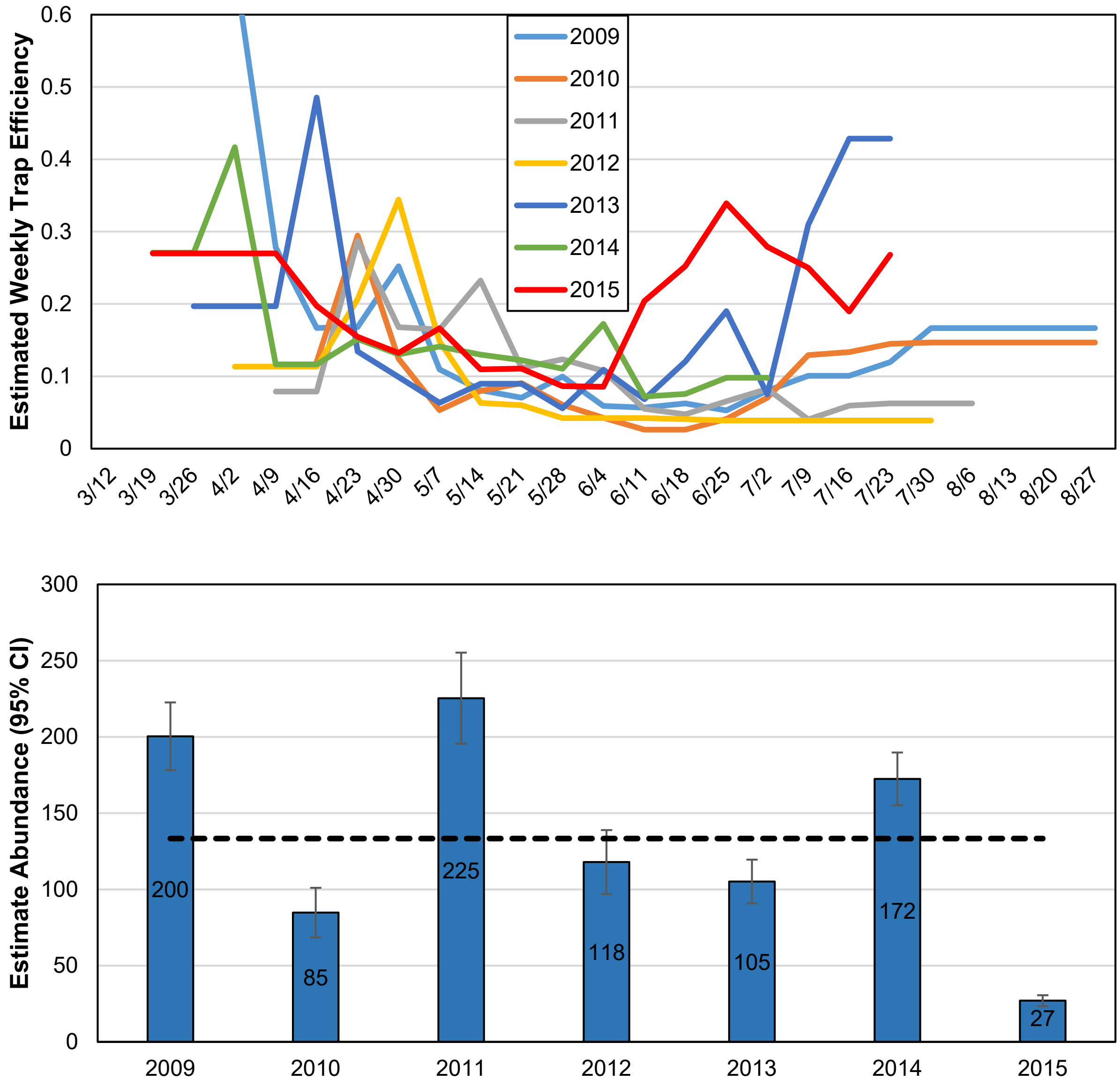
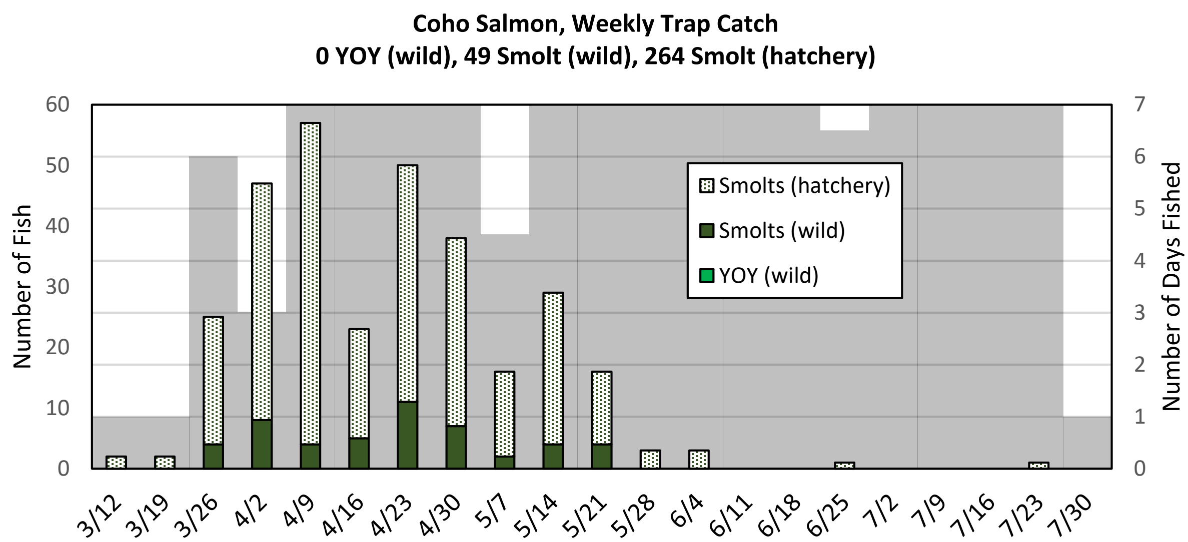


Figure 5.3.10. Estimated average weekly capture efficiency (upper panel) and population estimate of Chinook salmon smolts (x1000) produced from the Dry Creek watershed upstream of Westside Road smolt trap site (lower panel), 2009-2015. Dashed line is the seven year average abundance for all years combined.

Coho were the least abundant of the three salmonid species captured. Hatchery smolts dominated the coho catch with 249 individuals captured. Steelhead parr and smolt capture was highest in May (Figure 5.3.11).



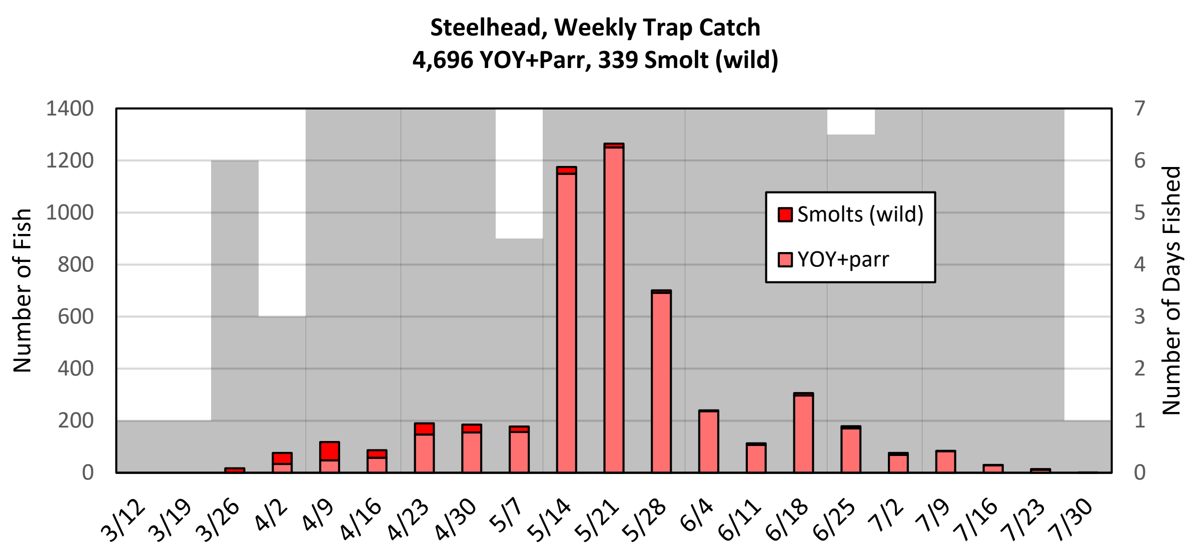


Figure 5.3.11. Weekly trap catch of juvenile coho salmon and steelhead in the Dry Creek rotary screw trap, 2015.

Coho smolt trap catch for the season was lower than the previous two years and similar to the catch in 2011 and 2012 (Figure 5.3.12). The capture of wild coho smolts was still quite low at 49 individuals and is similar to previous years totals. Steelhead smolt and parr captures (339 and 4,696) was similar to previous years totals.

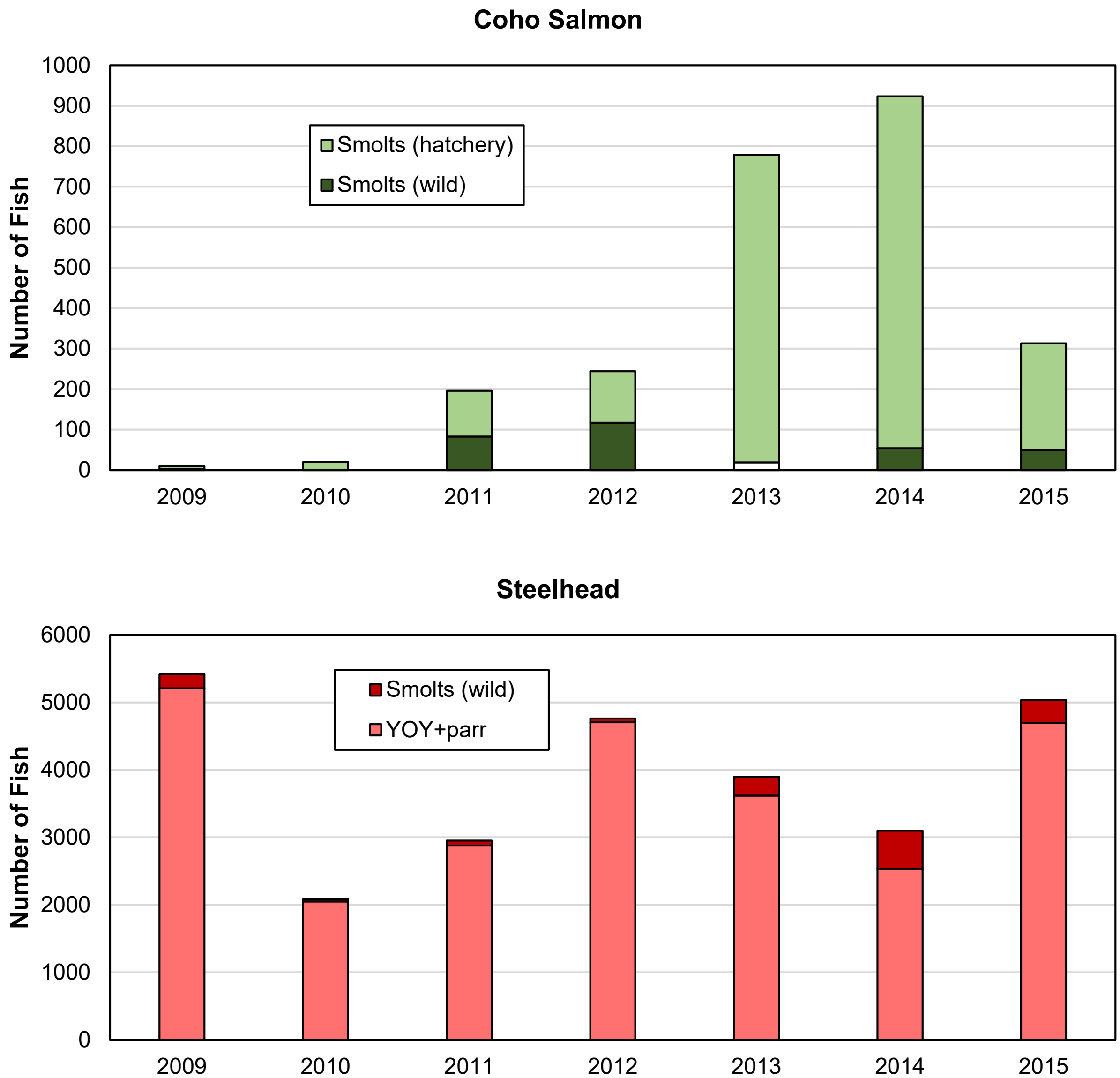


Figure 5.3.12. Trends in trap catch for coho smolts and steelhead smolts and juveniles, 2009-2015.

Weekly sizes of all salmonids captured at the Dry Creek trap increased in size over the course of the trapping season in 2015 (Figure 5.3.13).

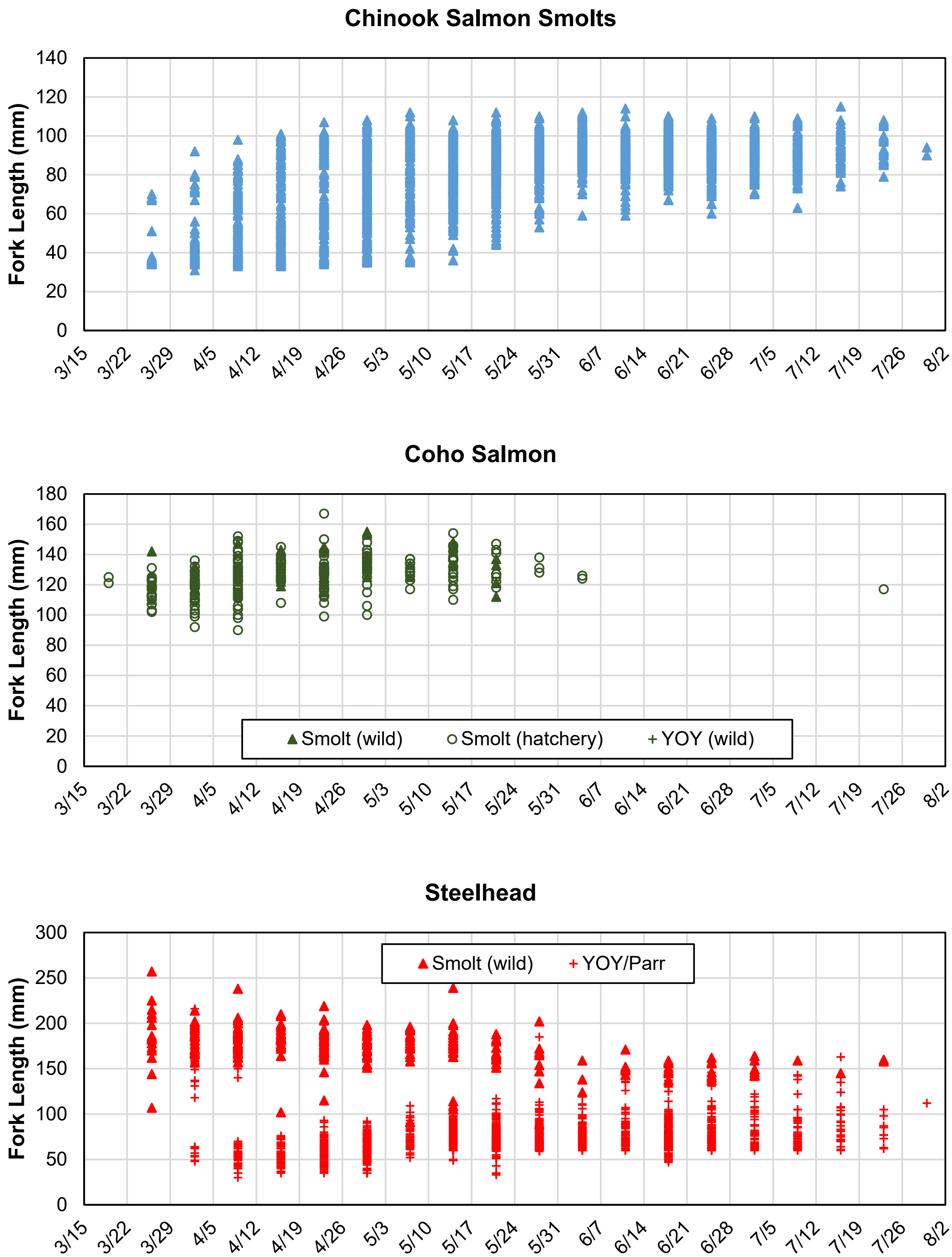


Figure 5.3.13. Fork lengths of juvenile salmonids captured in the Dry Creek rotary screw trap by week, 2015.

Conclusions and Recommendations

Because construction in the demonstration mile was not fully completed, we were unable to conduct biological monitoring in all features in both summer and winter until 2015. Our method for validating fish use in the late fall through winter has been through the use of PIT antennas within the backwaters. This approach provided data that various life stages of all three species are indeed using these habitats in the winter and we expect this method will continue to be useful as a way to document that use for future habitat enhancement phases. Unfortunately, marginal visibility due to high turbidity and vegetation growth in newly-created off-channel habitats hampered our ability to effectively observe fish during summer/fall snorkel surveys and these features are largely too deep to sample with a backpack electrofisher. However, sampling in areas adjacent to the off-channel features where rootwads and constructed riffles were implemented suggested that juvenile salmonids are finding and using those habitats. The CMP sampling framework proved useful as a way of understanding our site-level data in a broader context. We plan to build on those efforts in future years. In the future, we will consider alternative methods for estimating summer use of these habitats by juvenile salmonids including PIT-tagging/antennas and radio telemetry.

CMP sampling has proven useful for informing Biological Opinion related questions in Dry Creek. Based on run-timing information gathered from operation of a DIDSON that is part of our CMP monitoring, we observed several hundred Chinook salmon entering mainstem Dry Creek prior to December 31, 2014 and an above average number of Chinook redds in mainstem Dry Creek. Despite this, the 2015 Chinook smolt estimate in 2015 was the lowest since we began operating the downstream migrant trap on Dry Creek in 2009. We suspect the cause may be related to redd scour and low egg to fry survival from a 5,830 cfs event (as measured at the mouth of Dry Creek, USGS gage 11465350) in early to mid-December. Based on stream channel characteristics, estimates of stream velocities and sediment mobilization are consistent with that supposition. Although the primary targets of habitat enhancements in Dry Creek are juvenile coho and steelhead, understanding phenomenon such as high winter flows underscore the importance of accounting for those factors that influence juvenile populations and highlight the significance of providing high flow refugia for particularly vulnerable life stages.

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CHAPTER 6: Tributary Habitat Enhancements

Tributary Habitat Enhancement

One component of the reasonable and prudent alternative (RPA) identified in the Biological Opinion is the enhancement of salmonid rearing habitats in tributaries to Dry Creek and the Russian River. A total of ten potential tributary enhancement projects are listed in the Biological Opinion with the requirement that the Water Agency implement at least five of these projects by the end of year 3 of the 15 year period covered by the Russian River Biological Opinion. The five projects that the Water Agency intended to complete were 1) Grape Creek Habitat Improvement Project; 2) Willow Creek Fish Passage Enhancement Project; 3) Mill Creek Fish Passage Project; 4) Wallace Creek Fish Passage Project; and 5) Grape Creek Fish Passage Project. The Water Agency entered into agreements with the Sotoyome Resource Conservation District, now named Sonoma Resource Conservation District (RCD), to coordinate and implement two of these projects (the Grape Creek Habitat Improvement Project and Mill Creek Fish Passage Project), and with Trout Unlimited to provide funding towards the Willow Creek Fish Passage Enhancement Project. The Water Agency was also coordinating work with the Sonoma County Department of Transportation and Public Works to implement the Wallace Creek and Grape Creek Fish Passage Projects. After efforts to secure landowner access for the Mill Creek Fish Passage Project were unsuccessful, the Water Agency abandoned efforts on the Mill Creek Fish Passage Project and directed the Sotoyome Resource Conservation District to substitute the Crane Creek Fish Passage Project. The Water Agency also amended its agreement with the RCD to allow the RCD to oversee the implementation of the Grape Creek Fish Passage Project. The Wallace Creek Fish Passage Project, again after efforts to secure landowner access were unsuccessful, has been abandoned. The Water Agency is working with the National Marine Fisheries Service on an alternative as a substitute for the Wallace Creek Fish Passage Project.

Grape Creek Habitat Improvement

Phase 1

The Grape Creek Phase 1 portion of the project consisted of installing 8 complex log and boulder structures along a 1,200 foot reach of Grape Creek upstream of the Wine Creek Road Crossing (Figures 6.1 and 6.2). Implementation of this work took place in July and August of 2009. All areas where vegetation was disturbed by heavy equipment were replanted with native plants prescribed by restoration staff from the RCD. Additional plantings were also installed per the request of the California Department of Fish and Wildlife, and permission of the landowner, in areas outside the active construction area in an effort to eventually expand the width of the riparian area. A total of 248 native trees and shrubs were planted along this reach of the project. During 2011, maintenance and weeding of the plantings was conducted. General observations of the log structures during and after high creek flows of 2011-2012 have not shown any

Figure 6.1. Grape Creek – Phase 1. In-Stream Large Woody Debris Structure Example (2009 post construction).

Figure 6.2. Grape Creek – Phase 1. In-Stream Large Woody Debris Structure Example. December 2014 winter flows.

changes or failures in any of the Phase 1 reach structures. The first post-construction monitoring efforts occurred during the summer of 2011 (Figure 6.3). Riparian plantings were monitored and maintained in 2012 (Figure 6.4). Follow-up post-construction monitoring efforts were conducted during the summer of 2013 and 2014 (Figure 6.5). The next post-construction monitoring efforts are scheduled for the summer of 2015.

Figure 6.3. Grape Creek – Phase 1. 2011 Post-Construction Monitoring.

Figure 6.4. Grape Creek – Phase 1. February 2012.

Figure 6.5. Grape Creek – Phase 1. December 2014.

Phase 2

The Grape Creek Phase 2 portion of the project consisted of installing 9 complex log and boulder structures and 2 bank layback areas along a 700 foot reach of Grape Creek upstream of the West Dry Creek Road Crossing (Figure 6.6). Implementation of this work took place over two construction seasons, in 2009 and 2010. Construction began in early October 2009 and was cut short due to rain. Revegetation took place in January 2010. In February 2010, portions of one structure (Site 5) were removed as an emergency measure to avoid bank erosion on the opposite bank as a result of the structure's movement during high flows. Construction resumed in late August 2010, with heavy equipment work completed in the first week of September, and final touches placed on erosion control in early October. The remaining vegetation was installed in early 2011 when the soil is sufficiently moist. The first post-construction monitoring efforts occurred during the summer of 2011 (Figure 6.7). General observations of the log structures during and after high creek flows of 2011-2012 have not shown any changes or failures in any of the Phase 2 reach structures (Figures 6.8 and 6.9). Riparian plantings were monitoring and maintained in 2012. Follow-up post-construction monitoring efforts were conducted during the summer of 2013.

Figure 6.6. Grape Creek – Phase 2. Large Woody Debris and Bank Layback Example.

Figure 6.7. Grape Creek – Phase 2. 2011 Post-Construction Monitoring.

Figure 6.8. Grape Creek – Phase 2. February 2012.

Figure 6.9. Grape Creek – Phase 2. February 2012.

Figure 6.10. Grape Creek – Phase 2. December 2014.

Figure 6.11. Grape Creek – Phase 2. December 2014.

Willow Creek Fish Passage Enhancement Project

Willow Creek is a tributary to the lower Russian River that once supported an abundant subpopulation of coho salmon. The creek continues to support significant potential spawning and rearing habitat; however, access to that habitat is blocked by impassable road culverts and a shallow braided channel that passes through forested wetland. To implement the Willow Creek Fish Passage Enhancement Project, the Water Agency contributed $100,000 in funding to Trout Unlimited towards the removal of a complete barrier in Willow Creek. On October 19, 2010, the Water Agency's Board of Directors approved the funding agreement with Trout Unlimited for the Willow Creek Fish Passage Enhancement Project. The $100,000 in funding was provided by the Water Agency to Trout Unlimited on January 26, 2011. During the summer of 2011, construction was completed for the Willow Creek Fish Passage Enhancement Project (Figures 6.12 and 6.13).

Figure 6.12. Willow Creek Bridge Installation. September 2011.

Figure 6.13. Willow Creek Bridge Installation. September 2011.

Crane Creek Fish Passage Project

The Water Agency originally intended to implement the Mill Creek Fish Passage Project. The Mill Creek Fish Passage Project required landowner permission from two property owners in order to design and construct the project. One of the property owners was willing to enter into an agreement to allow the project to move forward; however, the second landowner gave multiple indications that they would allow the project to move forward, but ultimately failed to ever sign any access agreements to allow project design to move forward. Multiple attempts at obtaining the necessary permissions from this landowner were made by the Sotoyome Resource Conservation District and the National Marine Fisheries Service. Still seeing no progress with this landowner, the Water Agency directed the Sotoyome Resource Conservation District in December 2010 to abandon its efforts on the Mill Creek Fish Passage Project and instead implement the Crane Creek Fish Passage Access Project (Figures 6.14 and 6.15). The Crane Creek Fish Passage Access Project consists of the removal of a barrier to fish passage caused by a bedrock outcropping at the lower end of Crane Creek near its confluence with Dry Creek. The proposed project design developed by Prunuske Chatham, Inc., consisted of creating a series of step pools through the bedrock outcropping to create sufficient depth and flow to allow fish passage (Figure 6.16). Design approval was obtained from National Marine Fisheries Service and the landowners in September of 2011. Construction began on October 1, 2011 and was completed on October 18, 2011.

Figure 6.14. Crane Creek Fish Passage Access Project. Bedrock outcropping.

Figure 6.15. Crane Creek Fish Passage Access Project. Chiseling pools in bedrock outcropping.

Figure 6.16. Crane Creek Fish Passage Access Project. Expanded pools in bedrock outcropping (February 2012).

Grape Creek Fish Passage Project

The Grape Creek Fish Passage Project consists of the modification of a concrete box culvert where Grape Creek flows under West Dry Creek Road (Figure 6.17). As part of the permit review and design approval process, the National Marine Fisheries Service noted that the project design did not meet their maximum allowable 0.5-foot drop height for barrier passage. In October 2010, the Water Agency proposed re-designing the project to cut into the culvert bottom instead of placing curbs on top of the culvert bottom in order to meet the 0.5-foot maximum drop height requirement. Because the culvert-bottom is a structural portion of the bridge and culvert, cutting into the culvert bottom substantially increases the design complexity and costs of implementing the project. Between October 2010 and March 2011, the Water Agency coordinated with the Sonoma County Department of Public Works on the proposed re-design of the project. In April 2011, National Marine Fisheries Service indicated that the proposed re-design provided by the Sonoma County Department of Public Works was acceptable. Because of the increased complexity and cost, the revised project design was required to be put out to bid as a general construction contract, which required detailed project drawings and construction specifications. The Water Agency worked with a consultant through the Sotoyome Resource Conservation District to prepare the project construction drawings and specifications. Construction of the Grape Creek Fish Passage Project was completed in October of 2012 (Figures 6.18 and 6.19).

Figure 6.17. Grape Creek Fish Passage Project – Flat culvert invert proposed for modification.

Figure 6.18. Grape Creek Fish Passage Project – Newly Constructed October 2012.

Figure 6.19. Grape Creek Fish Passage Project – First Flows November-December 2012.

Mill Creek Fish Passage Project

The Water Agency had been working towards the construction of the Wallace Creek Fish Passage Project, which would have consisted of the modification of a concrete box culvert where Wallace Creek flows under Mill Creek Road. Engineering designs were completed and the National Marine Fisheries Service had approved those engineering designs for the project. The County of Sonoma Permit and Resource Management Department had submitted permit applications and coordinated site visits with California Department of Fish and Wildlife, National Marine Fisheries Service, U.S. Army Corps of Engineers, and the North Coast Regional Water Quality Control Board. Unfortunately, the Water Agency was been unable to secure the necessary landowner permissions from two of the three landowners in the project area. Because of the inability to secure the necessary landowner permission for the project, the Water Agency abandoned efforts to construct the Wallace Creek Fish Passage Project and began working with the National Marine Fisheries Service on an alternative as a substitute for the Wallace Creek Fish Passage Project.

In April of 2015, the National Marine Fisheries Service acknowledged that a proposal by the Water Agency to provide $200,000 in funding towards the construction of the Mill Creek Fish Passage Enhancement Project would meet the intent of the Russian River Biological Opinion and would be considered as the completion of the fifth and final tributary enhancement project required under the Russian River Biological Opinion. The Mill Creek Fish Passage Enhancement Project is a high-value project that would restore coho salmon access into 11.2 miles of upper Mill Creek. The initial estimate for the Mill Creek Fish Passage Enhancement Project described in the Russian River biological Opinion estimated the cost of the project at $100,000 to $200,000; however, recent estimates place the costs closer to $1,500,000. The Water Agency will donate $200,000 towards the project costs, which is consistent with the original estimate. The remaining funding for the project will come from NOAA grant funding and California Department of Fish and Wildlife Fisheries Restoration Grant Program funding. The project, which will be constructed in the summer of 2016, will allow for fish passage past an existing rock and mortar sill that is a barrier for fish passage under most flow conditions (Figure 6.20).

Figure 6.20. Mill Creek Fish Passage Project. Existing passage barrier in Mill Creek. December 2009.

CHAPTER 7 : Coho Salmon Broodstock Program Enhancement

The Biological Opinion and Consistency Determination require the Water Agency to increase production of coho salmon smolts from the Russian River Coho Salmon Broodstock Hatchery Program (Coho Program). The Coho Program is located at the Don Clausen Fish Facility (Warm Springs Hatchery) at the base of Lake Sonoma on Dry Creek. Initiated in 2001, this innovate program is a multi-partner effort involving USACE, CDFW, NMFS, University of California/California Sea Grant, and the Sonoma County Water Agency (SCWA). Native Russian River coho salmon and neighboring Lagunitas (Lagunitas and Olema) Creek coho salmon stock are bred according to a genetic matrix (provided by NMFS Southwest Fisheries Science Center) and progeny are released to more than 20 streams in the Russian River watershed. Fish are released in spring as fry, in fall as fingerlings, and during winter and early spring as smolts. The Biological Opinion requires USACE to fund most hatchery operations and monitoring, but also requires the Water Agency to provide resources to CDFW to produce a minimum of 10,000 coho smolts for release directly into Dry Creek.

The Water Agency purchased 15 tanks for the Coho Program in spring 2010 and they were installed by USACE in fall 2010. These tanks were operational by January of 2011, and have since been used to increase space for juvenile rearing, as well as for holding adult returns, and for the streamside imprinting tanks used on Dutch Bill Creek and Green Valley Creek. The Water Agency also hired a technician in spring 2010 and she has been working full time at the hatchery since the summer of 2010. The technician's primary duties at the hatchery include assisting the Coho Program Biologists with seasonal inventories of Broodstock. Starting in the summer of 2013 she began managing teams of SCWA program assistants on special projects; such as spawning, rearing, tagging and release of all coho salmon progeny.

The Water Agency's hatchery support technician continued to work with the biologists from the Coho Program throughout the 2013-14 release year. In addition to providing direct hatchery support, the technician was the lead point of contact for scheduling additional help for the Coho Program from available Water Agency Natural Resource Program Assistants (NRPA's). The Water Agency technician and the NRPA's primarily assisted the Coho Program with PIT-tagging efforts, juvenile releases, and the smolt imprinting efforts.

Beginning in 2014 and continuing in 2015, the Water Agency provided support to the hatchery on an as need basis as opposed to a full-time hatchery support technician. The primary role of the hatchery support was to assist with the PIT-tagging effort. Coordinated through the lead biologist at hatchery, the Water Agency would send 2 to 4 technicians per day during the tagging season to help complete this effort. This resulted in a substantial amount of hatchery support provided by the Water Agency during the 2015-16 release season.

After achieving the highest production total in the history of the program during the 2014-15 release year, the Coho Program experienced one of its lowest juvenile production totals the very next year. Due to the drought, lower lake levels and warmer water temperatures at the hatchery in 2014 (Figure 7.1) affected the egg development of the female broodstock during the 2014-15 spawning season. This led to very low eye-up rates and hatch rates (52% and 49%, respectively), which in turn resulted in a juvenile production total of only 72,721 for the 2015-16 release year (Table 7.1).

The monitoring component of the Coho Program relies, in part, on detections of PIT-tagged adult coho returning to the Russian River Watershed. By applying the fractional marking rate of hatchery releases for a given cohort to the number of fish detected on the Duncans Mills PIT antenna array in the Russian River estuary (after adjusting for PIT antenna detection efficiency), an expanded count of hatchery returns can be estimated. Typically, that fractional marking has been 15%. however, because of the low juvenile production resulting from the 2014-15 spawning season and in order to achieve a higher likelihood of detecting adult returns, the Coho Program's Technical Advisory Committee decided to double the proportion of fish PIT-tagged from 15% to 30% for the 2015-16 release season (Table 7.1). As in past years, Water Agency technicians provided a significant amount of labor in order to complete the PIT-tagging effort for the 2015-16 release year. Due to the reduced number of fish available for release, though, many fewer release streams were stocked. However, Dry Creek still received approximately 10,000 smolts released during spring 2016. The Dry Creek release group was split into 3 sub-groups and released approximately 2 weeks apart from one another during April and May, 2016.

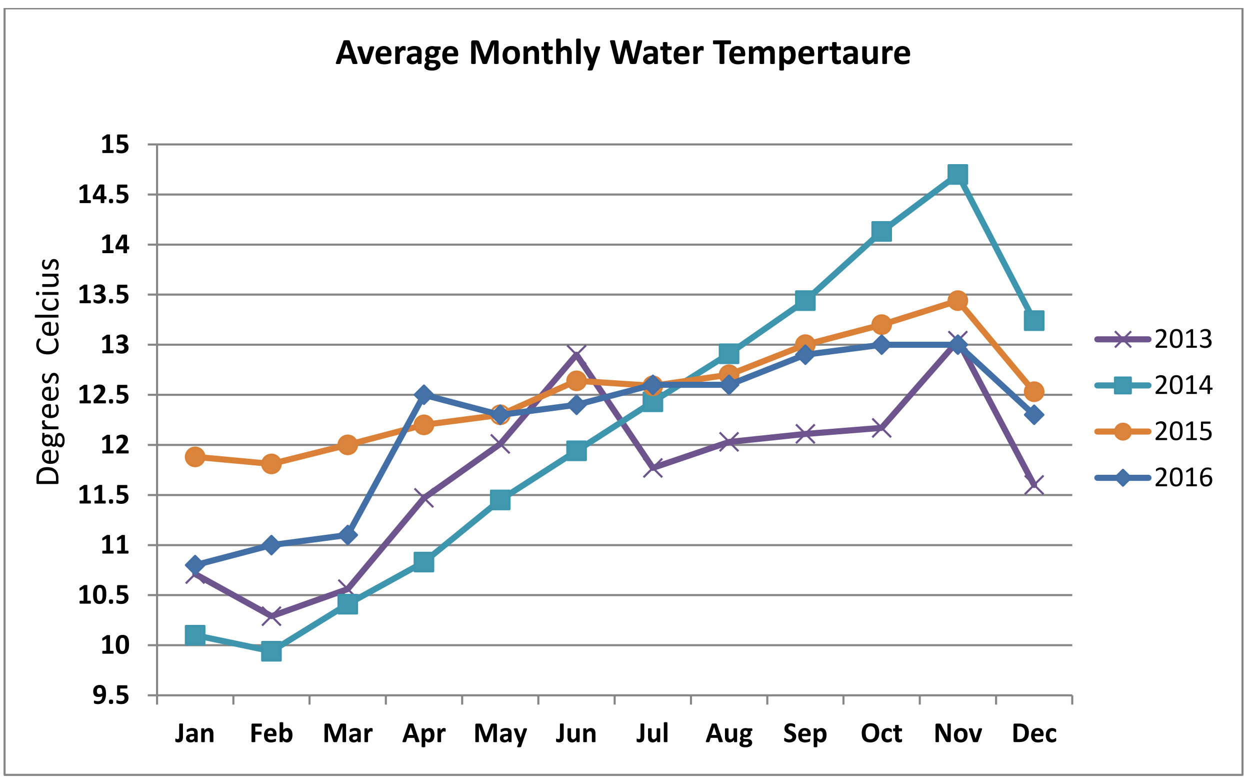

Figure 7.1. Russian River Coho Program average monthly water temperature from 2013-2016 (B. White, USACE, personal communication).

Table 7.1. Russian River Coho Program 2015-16 juvenile releases (B. White, USACE, personal communication).

Release Date	Release Stream	# Released	Ave FL (mm)	Ave Wt. (g)	Tagging Strategy
6/17/2015	Dutch Bill Creek	1,008	70 ± 5	4.1 ± 1.0	CWT + 100% PIT
6/18/2015	Mill Creek	509	67 ± 3	3.5 ± 0.6	CWT + 100% PIT
6/18/2014	Green Valley Creek	305	68 ± 4	3.8 ± 0.6	CWT + 100% PIT
2015 Spring Release Total:		***1,822***			
11/19/2015	Gray Creek	4,021	90 ± 11	9.3 ± 3.7	CWT + 30% PIT
11/20/2015	Gilliam Creek	4,107	87 ± 7	8.1 ± 2.2	CWT + 30% PIT
11/20/2015	Walker Creek (Marin)	2,211	88 ± 6	8.1 ± 1.6	CWT only
11/25/2015	Mill Creek	8,969	89 ± 7	8.7 ± 2.0	CWT + 30% PIT
12/7/2015	Willow Creek	9,032	91 ± 7	9.2 ± 2.0	CWT + 30% PIT
12/9/2015	Green Valley Creek	8,989	93 ± 7	9.6 ± 2.3	CWT + 30% PIT
12/10/2015	Dutch Bill Creek	8,989	94 ± 7	10.0 ± 2.3	CWT + 30% PIT
2015 Fall Release Total:		***46,318***			
4/15/2016	Dry Creek Grp.1	3,322	115 ± 9	17.1 ± 4.4	CWT + 30% PIT
4/18/2016	Dutch Bill Creek Grp.1*	1,675	110 ± 9	15.2 ± 3.5	CWT + 30% PIT
4/18/2016	Green Valley Creek Grp.1	2,536	112 ± 8	16.1 ± 3.5	CWT + 30% PIT
4/25/2016	Mill Creek Grp.2	2,384	115 ± 9	17.4 ± 4.5	CWT + 30% PIT
5/2/2016	Dry Creek Grp.2	3,318	116 ± 9	17.3 ± 4.2	CWT + 30% PIT
5/2/2016	Green Valley Creek Grp.2	2,328	114 ± 9	16.8 ± 4.4	CWT + 30% PIT
5/3/2016	Dutch Bill Creek Grp.2*	1,705	113 ± 8	15.9 ± 3.6	CWT + 30% PIT
5/4/2016	Mill Creek-Pond**	2,391	110 ± 8	15.8 ± 3.3	CWT + 30% PIT
5/13/2016	Dry Creek Grp.3	3,284	117 ± 9	17.7 ± 4.3	CWT + 30% PIT
5/19/2016	Dutch Bill Creek Grp.3*	1,638	116 ± 8	17.5 ± 4.0	CWT + 30% PIT
2016 Smolt Release Total:		***24,581***			
2015-16 Release Total:		***72,721***			

* Imprinted for 13 days in stream-side tank prior to release.

** Imprinted for 27 days in flashboard dam pond prior to release.

CHAPTER 8: Wohler-Mirabel Water Diversion Facility

Introduction

The Water Agency diverts water from the Russian River to meet residential and municipal demands. Water is stored in Lake Sonoma and Lake Mendocino, and releases are made to meet downstream demands and minimum instream flow requirements. The Water Agency's water diversion facilities are located near Mirabel and Wohler Road in Forestville. The Water Agency operates six Ranney collector wells (large groundwater pumps) adjacent to the Russian River that extract water from the aquifer beneath the streambed. The ability of the Russian River aquifer to produce water is generally limited by the rate of recharge to the aquifer through the streambed. To augment this rate of recharge, the Water Agency has constructed several infiltration ponds. The Mirabel Inflatable Dam (Inflatable Dam) raises the water level and allows pumping to a series of canals that feed infiltration ponds located at the Mirabel facility. The backwater created by the Inflatable Dam also raises the upstream water level and submerges a larger streambed area along the river. Three collectors wells, including the Agency's newest and highest capacity well, are located upstream of Wohler Bridge. These wells benefit substantially from the backwater behind the Dam.

Mirabel Fish Screen and Ladder Replacement

To divert surface water from the forebay of Mirabel Dam, The Water Agency operates a pump station on the west bank of the river. The pump station is capable of withdrawing 100 cfs of surface flow through two rotating drum fish screens in the forebay. The fish screens have been functioning since the dam was constructed in the late 1970's. However, they fail to meet current velocity standards established by NMFS and CDFW to protect juvenile fish. The Biological Opinion requires the Water Agency to replace the antiquated fish screens with a structure that meets modern screening criteria. In 2009, the Water Agency employed the engineering firm of Prunuske Chatham, Inc. to prepare a fish screen design feasibility study. The report was completed in December 2009.

The feasibility study was conducted to develop a preferred conceptual design that meets many of the project objectives while ensuring that the fish screening facilities adhere to contemporary fish screening design criteria. A Technical Advisory Committee composed of the Water Agency engineering and fisheries biologist staff, NMFS, and CDFW provided guidance in refining the objectives and identifying alternatives. Six concept alternatives were evaluated for meeting the project objectives. Schematic designs and critical details were developed for these concept alternatives to assess physical feasibility and evaluate alternatives relative to the objectives. The preferred concept design alternative was determined through an interactive evaluation and was selected because it meets or exceeds the project objectives.

In 2010, the Water Agency solicited qualifications from engineering firms, and a list of qualified consultants was created from the responses. The Water Agency selected HDR Engineering

(HDR) because of its demonstrated experience with this type of work and the strength of their proposed project manager, who has a proven track record with fish passage and screening projects. The Water Agency and HDR entered into an Agreement for Engineering Design Services for the Mirabel Fish Screen and Fish Ladder Replacement Project in June of 2011. In 2011 and 2012, HDR completed work on preliminary engineering, geotechnical analysis, hydraulic modeling, development of construction drawings and specifications. HDR's final construction drawings and specifications are anticipated in early 2013. HDR will also provide engineering support during bidding and construction. HDR's design process included consultation at different design steps with the Technical Advisory Committee described above.

Because the fish ladder enhancement identified in the feasibility study is not required by the Biological Opinion, the Water Agency applied for funds from CDFW's Fishery Restoration Grant Program (FRGP) in 2010 to help defray costs associated with fish ladder design. The Director of CDFW awarded the grant to the Water Agency in February 2011. The Water Agency also submitted a second application for FRGP funds in 2012 to help defray costs associated with fish ladder construction. In February of 2013, CDFW approved $1,184,049.00 in FRGP funds towards the construction of the new fishway at Mirabel to improve fish passage at the facility.

In January 2013, the Water Agency's Board of Directors approved and adopted an Initial Study and Mitigated Negative Declaration in accordance with the California Environmental Quality Act (CEQA).

The CEQA document for the project provided a discussion of potential environmental impacts related to the construction, operation, and maintenance of the proposed fish screen and fish ladder modifications. Project construction activities require isolating the work area from the active flow of the Russian River, demolishing the existing fish screen/intake and fish ladder structures on the western bank of the Russian River, and constructing the new fish screen and fish ladder structures. The new facilities will extend approximately 40 feet farther upstream and approximately 100 feet farther downstream than the existing facilities. This larger footprint is necessary to meet contemporary fish screen and fish passage design criteria. Figure 8.1 shows a plan view of the project design. Figure 8.2 shows a conceptual design drawing of the project components.

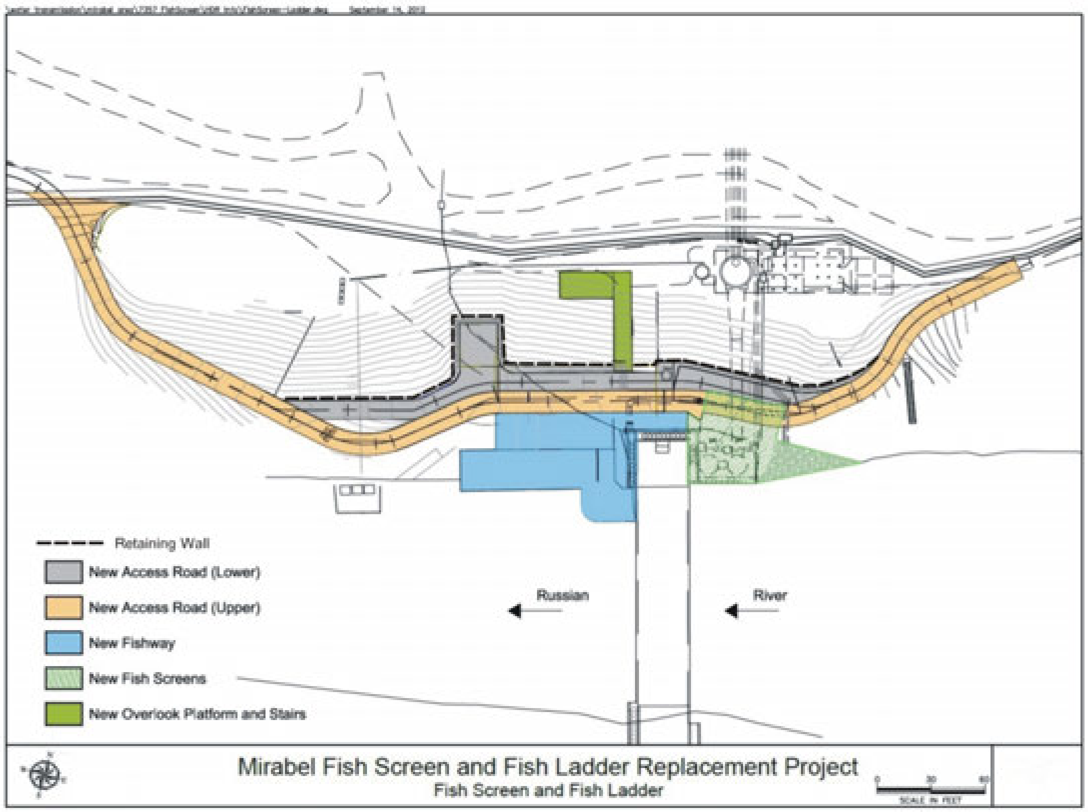

Figure 8.1. Conceptual plan view drawing of new fish screen and fishway structure at Mirabel.

Figure 8.2. Artist rendering of new fish screen and fishway structure at Mirabel.

Fish Screen

The proposed intake screen will consist of six 12-foot tall by 6-foot wide panels, with a total area of 432 square feet. The new fish screen will also incorporate a cleaning system to ensure that the screen material does not become clogged. Clogged screens result in higher flows through unclogged portions of the screen, which can lead to fish getting trapped against the screen. The cleaning mechanism is anticipated to be an electric motor-driven mechanical brush system that periodically moves back and forth to clean the intake screen structure.

Fish Ladder

A vertical slot type fish ladder was selected as the recommended design to provide passage for upstream migrating salmonids. Vertical slot fish ladders are commonly used for salmon and steelhead (among other fish species) throughout the world. A vertical slot fish ladder consists of a sloped, reinforced concrete rectangular channel separated by vertical baffles with 15-inch wide slots that extend down the entire depth of the baffle. The baffles are located at even increments to create a step-like arrangement of resting pools.

The design will be self-regulating and provide consistent velocities, flow depths, and water surface differentials at each slot throughout a range of operating conditions. It is anticipated that the ladder will be configured to accommodate a range of fish passage conditions while the Mirabel Dam is up and river flows ranging from 125 to 800 cubic feet per second. Fish passage while the Mirabel Dam is down will also be accommodated, but is not the primary focus of design. The fish ladder will extend approximately 100 feet further downstream than the existing fish ladder at the site.

Fisheries Monitoring Components

The Water Agency currently conducts a variety of fisheries monitoring activities at its Mirabel Dam facilities. The new fish ladder design will support these monitoring activities by providing a dedicated viewing window and video equipment room and a fish trapping and holding area built into the fish ladder. The monitoring information collected by Water Agency staff is critical in tracking population trends and movement of different species in the Russian River system.

Education Opportunities

The existing facility at Mirabel is visited every year by approximately 3,000 schoolchildren as part of the Water Agency’s water education efforts. The existing facility allows schoolchildren to see a critical component of the Water Agency’s water supply system, but the views of the top of the existing fish ladder do not offer much opportunity for observing and learning about the fisheries of the Russian River system. The project includes a viewing area, separate from the video monitoring viewing window, which will allow visitors to see into the side of the fish ladder. The educational experience for schoolchildren will be improved by having the opportunity to actually see fish travelling up or down the fish ladder.

Supporting Components

The project design includes a variety of other components that support the primary fish screen and fish ladder aspects of the project. These other components consist of items such as seismic stabilization of the soils around the Mirabel dam, replacement of the buoy warning line upstream of the Mirabel Dam, modification of the existing access road to the project site, and the installation of a viewing platform to allow visitors a safe location to view the overall facility. The existing access road down to the Mirabel Dam is a steep one-way road. Vehicles going down to the Mirabel Dam area must turn around or back up the road down to the project site. The proposed project includes a modification of the access road so that the road will not be as steep and will include both an entrance and exit ramp from the Mirabel Dam site. A stairway from the top of bank down to the Mirabel Dam will allow visitor access from the upper levee road area down to the Mirabel Dam.

Construction Status

In March 2014, Hayward Baker began construction on the first phase of site improvements at the Mirabel Dam. This work consisted of the seismic stabilization of the soil area around the area of the Mirabel intake screens and fish ladder on the west bank of the Russian River. Seismic stabilization consisted of the installation of approximately 300 compacted stone columns along the levee berm at the Mirabel facility. The Mirabel seismic improvement work was completed in July of 2014 by Hayward Baker, which then allowed the second phase of construction activities to begin. Once Hayward Baker had demobilized their equipment from the work area, a second contractor (F&H) mobilized to the site in July of 2014 to begin the construction of the fish screen, fish ladder, and viewing chamber project. By the end of 2014, demolition of the original intake structure and fish ladder was complete. At the beginning of 2015, high flows in the Russian River resulted in a temporary shut-down of construction activities (Figure 8.3); however, by mid-January 2015, construction activities were once again underway and continued through 2015 (Figure 8.4 through 8.11).

Figure 8.3. Russian River high flows shut down work progress at the job site. December 12, 2014.

Figure 8.4. High flows in the Russian River had receded, the job site was again dewatered, and work was under way again by January 12, 2015.

Figure 8.5. Sheet pile retaining wall for new structure was complete April 29, 2015.

Figure 8.6. In May of 2015, pile-driving of support piles for the new structure was underway.

Figure 8.7. Towards the end of June 2015, installation of the temporary cofferdam just upstream of the Wohler Bridge began again. July 8, 2015 photo.

Figure 8.8. By August of 2015, construction of the new fish ladder, screens, and viewing gallery was progressing.

Figure 8.9. The outlet structure walls of the new fishway have taken shape. November 10, 2015.

Figure 8.10. New vertical slot fishway. Photo showing fishway channel with the rebar framework of the vertical slot baffles. Photo also shows the openings for the viewing windows into the side of the fishway. December 2015.

Figure 8.11. Upstream entrance of new vertical slot fishway. Scaffolding visible on the right side of photo is the location of where the new vertical panel fish screens will be located. December 2015.

CHAPTER 9 : Adult Salmonid Returns

Adult Salmonid Escapement

Since 2000, the Water Agency has been operating video cameras in the east and west fish ladders to assess the adult Chinook salmon run passing the Mirabel inflatable dam located at river km 39 (rkm 39). In 2014 and 2015, however, construction of a new fish ladder and fish screens at Mirabel prevented inflation of the dam which, in turn, prevented us from implementing the video monitoring system in the fish ladders. As an alternative, the Water Agency adjusted its sampling program by installing and operating (1) a video camera in the Russian River at the Healdsburg dam fish ladder just upstream of the Dry Creek confluence (rkm 55) and (2) a DIDSON camera in Dry Creek (USGS, rkm 0.36) just upstream of the confluence with the Russian River (Dry Creek/Russian River confluence at rkm 52). Because little Chinook spawning habitat exists between either of these locations and the Mirabel dam, conceptually, accurate counts of adult Chinook at these sites should represent the majority of the run.

Methods

An underwater digital camera and lighting system was installed in a fishway pool (“camera pool”) near the upstream end of the Healdsburg fish ladder and video was recorded to a hard drive located in a nearby building. The passage of adult salmonids through the Healdsburg fish ladder was assessed from September 15 until December 9, 2015 when high stream flows forced the removal of the camera. Individuals were counted as moving upstream once they exited the upstream end of the camera pool. For each adult salmonid observed, the reviewer recorded the species, date, and time of upstream passage. During periods of low visibility it was not always possible to identify fish to species, although identification as an adult salmonid was usually possible. Adult salmonids that could not be identified to species were lumped into a general category called “unknown salmonid.” Unknown salmonids were then partitioned into species by taking the proportion of each species positively identified in the ladder on a given day and multiplying the number of unknown salmonids on that same day by these proportions. On days when no salmonids could be identified to species, an average proportion from adjacent days was used to assign species to the unidentified salmonids on that day.

Data collection in Dry Creek using a DIDSON was funded from a Fisheries Restoration Grant awarded to the Water Agency for implementation of the Coastal Monitoring Program (CMP) in the Russian River. Because species identification is not possible from DIDSON, we attempted to use identify species on a video camera paired with the DIDSON and use that information to assign species to fish observed on the DIDSON but not observed on the video camera. Because of the low sample size resulting from too few fish swimming close enough to the video camera to identify, this approach proved infeasible. Instead we relied on fish size, which can be reliably estimated with the DIDSON, to assign fish with body size of 2 feet or greater as a salmonid. Next, based on historical run-timing at Mirabel (years 2008-2013), we further apportioned salmonids counted prior to January 23, 2016 as Chinook, steelhead or coho. Finally, beginning January 23, 2016 all adult salmonids were assumed to be steelhead.

Results

In 2015 the Healdsburg fish ladder digital video system operated nearly continuously from September 15 to December 9. For the majority of the monitoring period, the image quality of video footage was sufficient to identify species and count fish passing through the fish ladder. The Dry Creek DIDSON operated from September 1, 2015 to April 16, 2016 except for brief periods when the DIDSON camera lost power or connection to the computer.

Chinook Salmon

During the monitoring period at Healdsburg, we arrived at 512 adult Chinook from a combination of positive identification of 428 individuals and proration of 83 individuals out of the 88 initially identified as “unknown salmonids”. A total of 42 Chinook were positively identified on an underwater video camera which was operated in Dry Creek alongside the DIDSON camera. In addition to these known Chinook, we observed 12,802 fish with a length greater than or equal to 2 feet on the USGS DIDSON camera. Based on their size we assumed all of these fish were adult salmonids (however, this assumption may not be valid – see ‘Conclusions and Recommendations’ section). Using historical run timing information from Mirabel, 2,467 of these 12,802 unknown salmonids were prorated to Chinook; the remainder were likely steelhead and coho. For the reasons discussed below, the sum of Chinook counts (3,020) from Healdsburg and Dry Creek is not necessarily comparable to minimum counts for other years from the Mirabel fish ladder; however, it is within the range of counts from previous years (Table 9.1). By combining historical information from Mirabel with DIDSON and video data from 2015 we were able to make inferences about the Chinook run across a similar time frame that the Mirabel video camera is typically operated (Figure 9.1). The Chinook run in 2015 began to ramp up in mid-November, and, based on Dry Creek DIDSON data, likely peaked in December.

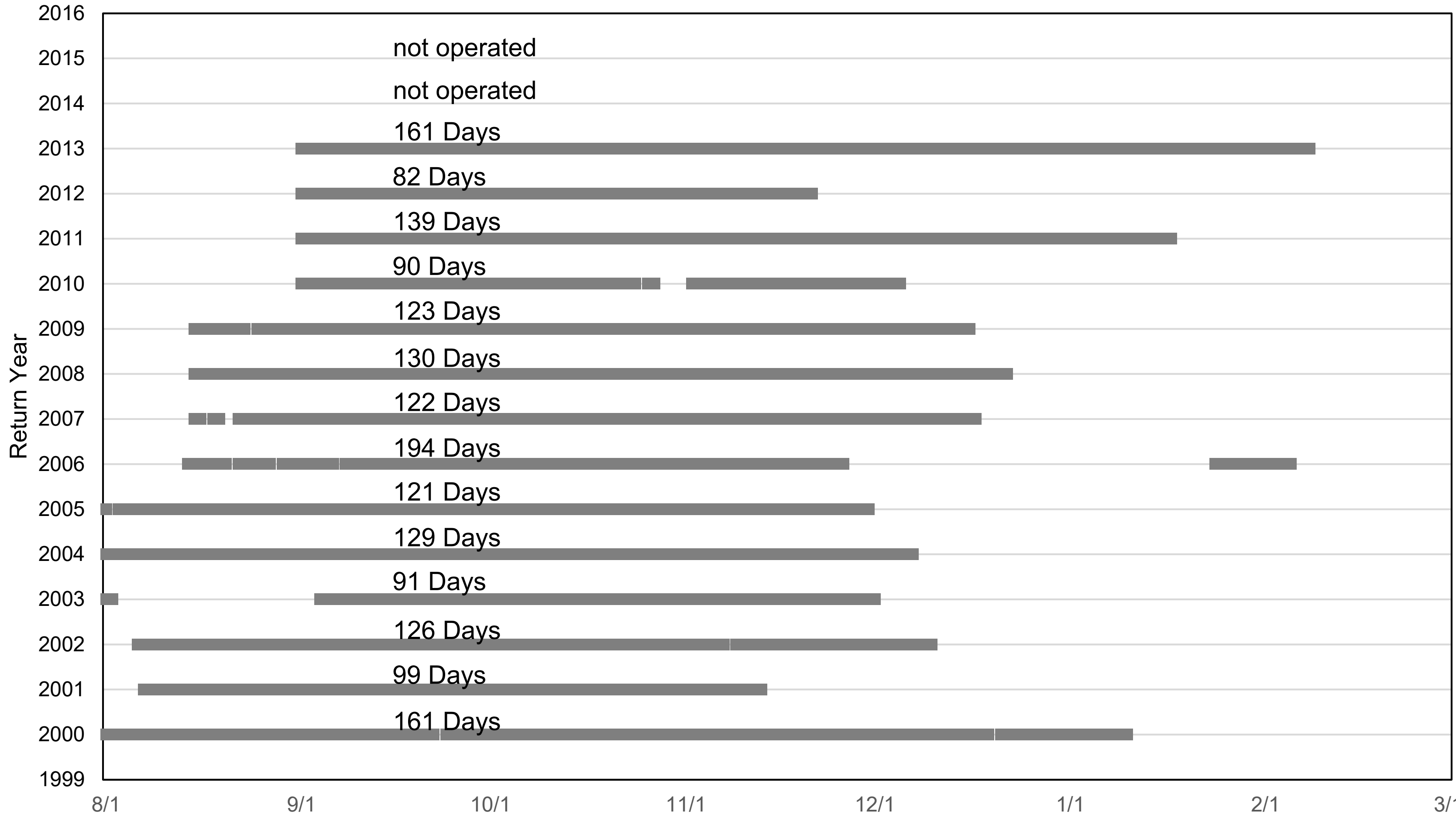

Figure 9.1. Period of operation by adult salmonid return year of video counting station at the Mirabel dam. 'Days' refer to the number of days of operation each year.

Table 9.1. Weekly count of adult Chinook salmon at the Mirabel Dam fish ladders, 2000-2015. Dashes indicate that no sampling occurred during that week.

Week	2000	2001	2002	2003	2004	2005	2006[1]	2007	2008	2009	2010	2011	2012	2013	2014[2]	2015[2]
15-Aug	0	0	1	--	0	0	0	0	0	0	--	--	--	--	not operated	not operated
22-Aug	1	0	8	--	0	1	1	0	0	0	--	--	--	--		
29-Aug	0	3	7	2	1	4	0	0	1	0	0	0	0	1		
5-Sep	9	1	18	7	1	4	0	0	0	0	0	0	1	1		
12-Sep	36	7	19	20	3	14	3	0	2	0	0	0	2	2		
19-Sep	25	12	65	23	8	14	4	1	17	0	3	1	0	1		
26-Sep	50	17	1223	181	16	31	8	4	84	0	1	158	70	17		
3-Oct	31	240	113	146	42	27	317	10	126	78	669	534	51	44		
10-Oct	115	51	628	515	52	112	87	39	82	562	896	390	551	4		
17-Oct	81	10	272	232	651	556	532	26	13	177	153	1070	1886	8		
24-Oct	465	300	153	532	2287	309	114	106	22	285	280	273	996	27		
31-Oct	64	661	505	2969	185	613	1531	250	511	135	94	223	1654	315		
7-Nov	23	81	2337	1289	1189	699	298	429	174	335	169	90	619	731		
14-Nov	182	--	20	47	221	127	459	154	15	38	43	120	851	1063		
21-Nov	201	--	37	95	57	63	53	96	24	129	113	266	50	179		
28-Nov	110	--	14	45	60	33	--	425	19	24	76	6	--	99		
5-Dec	19	--	53	--	16	--	--	476	18	9	5	1	--	172		
12-Dec	15	--	--	--	--	--	--	4	8	28	--	2	--	125		
19-Dec	17	--	--	--	--	--	--	--	13	--	--	10	--	73		
26-Dec	1	--	--	--	--	--	--	--	--	--	--	16	--	32		
2-Jan	0	--	--	--	--	--	--	--	--	--	--	2	--	53		
9-Jan	0	--	--	--	--	--	--	--	--	--	--	10	--	58		
16-Jan	--	--	--	--	--	--	--	--	--	--	--	1	--	28		
23-Jan	--	--	--	--	--	--	0	--	--	--	--	--	--	73		
30-Jan	--	--	--	--	--	--	0	--	--	--	--	--	--	36		
6-Feb	--	--	--	--	--	--	--	--	--	--	--	--	--	10		
Total	1,445	1,383	5,474	6,103	4,788	2,607	3,407	2,021	1,129	1,800	2,502	3,173	6,730	3,152	--	--

1 Video cameras were reinstalled and operated from 4/1-6/27, 2007 but no Chinook salmon were observed.
2 Video cameras not operated in 2014 and 2015 because this site was under construction in order to construct the new fish screens and ladder.

Coho Salmon

During the monitoring period for the 2015 return year, we observed 20 adult coho passing the Healdsburg fish ladder. These images were reviewed by fisheries biologist from the Water Agency, NMFS, and California Sea Grant (CSG). Because of the timing of camera operations, which are tied to dam operations, and the location of these monitoring sites upstream of significant amounts of coho habitat in the basin, these counts are not the best indicator of adult coho returns to the basin. Instead, we suggest the basinwide spawner survey estimate of 160 redds (95% CI: 89-231) as the most comprehensive and accurate indicator of all adult coho (hatchery- and natural-origin) returning to the Russian River basin in 2015-16. This estimate is based on spawner surveys in the coho stratum of the Russian River Coastal Monitoring Program sample frame (see Adams et al. 2011 for details).

Steelhead

Based on hatchery returns, steelhead migrate and spawn in the Russian River primarily between December and March; however, we removed the Healdsburg fish ladder camera in early December. Without steelhead counts after early December, it is impossible to arrive at an accurate basinwide steelhead count; therefore, the 2015-16 data collected at the Healdsburg dam are of little value (a total of 3 were observed on the Healdsburg Camera).

Conclusions and Recommendations

By establishing a fish counting station on mainstem Dry Creek and mainstem Russian River at Healdsburg, estimation of the percentage of Chinook salmon utilizing Dry Creek as compared to the mainstem Russian upstream of Dry Creek should be possible. However, technical and environmental issues challenged our ability to make an accurate assessment for the 2015-16 return year. These challenges lead us to suggest caution when comparing these numbers to video counts at the Mirabel dam.

From a technical standpoint, locating the Healdsburg fish ladder camera in a manner that allowed for a complete census of fish migrating through the camera pool was problematic. Unlike the Mirabel fish ladders where a specially-constructed camera box forces fish to swim in front of the camera, at the Healdsburg fish ladder it was possible for a Chinook to jump over the cameras field of view and avoid detection.

Though DIDSON is noted for its utility in allowing detection of large-bodied fish such as adult salmonids even when turbidity is high, it does not allow for distinguishing species. This presents a problem in Dry Creek where run-timing for Chinook, coho and steelhead overlap. To overcome this issue, we estimated a run-timing end date of January 22, 2016 for Chinook based on historical Chinook run-timing data from the Mirabel video fish counting station. Though Chinook run-timing certainly does not conform to a specific calendar date, a sensitivity analysis suggested that, in most years, choosing an “incorrect” migration termination date would only slightly affect our count. Another potential source of error is our assumption that all fish greater than 2 feet were adult salmonids when, in fact, this may not be true. We are aware that large-bodied non-salmonids are present in the mainstem Russian River (e.g., Sacramento pikeminnow) which could inflate our count of adult salmonids. These issues illustrate an important limitation of DIDSON monitoring that could be particularly problematic in systems like Dry Creek where there is a high and/or variable degree of overlap in run-timing among species and where large-bodied non-salmonids are present.

A significant environmental challenge faced by migrating adult Chinook in 2015 was the closed estuary and low flow condition prevalent for a significant portion of the sampling period. Upstream-migrating salmonids had relatively few opportunities to enter the Russian River during the historical Russian River Chinook migration window, and those that did faced exceptionally low streamflows in 2015. The mouth of the estuary closed on September 7 (based on water surface elevations) and remained closed until it was artificially breached on October 4. The mouth re-closed on October 10 and remained closed until November 5 followed by periodic closures through mid-December. The number of adult Chinook observed in 2015-16 illustrates that the Chinook population in the Russian can overcome these environmental obstacles through flexibility in run-timing.

Chinook Salmon Spawning Ground Surveys

Although not an explicit requirement of the Biological Opinion, the Water Agency performs spawning ground surveys for Chinook salmon in the mainstem Russian River and Dry Creek. This effort compliments the required video monitoring of adult fish migration and has been stipulated in temporary D1610 flow change orders issued by the State Water Resources Control Board to satisfy the Biological Opinion (see Pursue Changes to D1610 flow chapter of this report). The Water Agency began conducting Chinook salmon spawning surveys in fall 2002 to address concerns that reduced water supply releases from Coyote Valley Dam (Lake Mendocino) may affect migrating and spawning Chinook salmon (Cook 2003). Spawner surveys in Dry Creek began in 2003.

Background information on the natural history of Chinook salmon in the Russian River is presented in the 2011 Russian River Biological Opinion annual report (SCWA 2011). The primary objectives of the spawning ground surveys are to (1) characterize the distribution and relative abundance of Chinook salmon spawning sites, and (2) compare annual results with findings from previous study years.

Chinook salmon spawner surveys were restricted to one reach of the Russian River and Dry Creek during 2015. A late-season spawning run of Chinook salmon coupled with heavy rainfall and subsequent high river flows in early December 2015 prevented field studies from being conducted during the peak migration period of salmon in the Russian River mainstem. Spawner surveys were possible in Dry Creek due to regulated, clear water releases from Lake Sonoma during fall 2015.

Methods

Chinook salmon redd (spawning nest) surveys are conducted annually in the Russian River during fall. Typically, the upper Russian River basin and Dry Creek are surveyed (Figure 9.2). The study area includes approximately 114 km of the Russian River mainstem from Riverfront Park (40 rkm), located south of Healdsburg, upstream to the confluences of the East and West Forks of the Russian River (154 rkm) near Ukiah. River kilometer (rkm) is the meandering stream distance from the Pacific Ocean upstream along the Russian River mainstem and for Dry Creek the distance from the confluence with the Russian River upstream. In 2003, the study area was expanded to include 22 rkm of Dry Creek below Warm Springs Dam at Lake Sonoma to the Russian River confluence.

The Chinook salmon spawning ground study consisted of a single-pass survey during the estimated peak of Chinook salmon fall spawning. A crew of two biologists in kayaks visually searched for redds along the streambed. Riffles with several redds were inspected on foot. The locations of redds were recorded using a global positioning system (GPS). Surveys were cancelled or postponed if increased turbidity from heavy rainfall obscured the detection of redds. Also, in recent years releases of highly turbid water from Lake Mendocino have prevented an accurate count of redds in Ukiah reach.

As mentioned above, Chinook salmon spawner surveys were curtailed during fall 2015 due to poor survey conditions. The Alexander Valley reach of the Russian River study area was surveyed on December 8, 2015. To follow salmon spawning period and determine the peak of spawning activity bi-monthly surveys were completed along Dry Creek from November 2015 to January 2016 (SCWA 2016). The survey conducted on December 16, 2015 along Dry Creek contained the largest weekly count of redds and was selected as the single-pass visit to represent of the abundance of redds in Dry Creek.

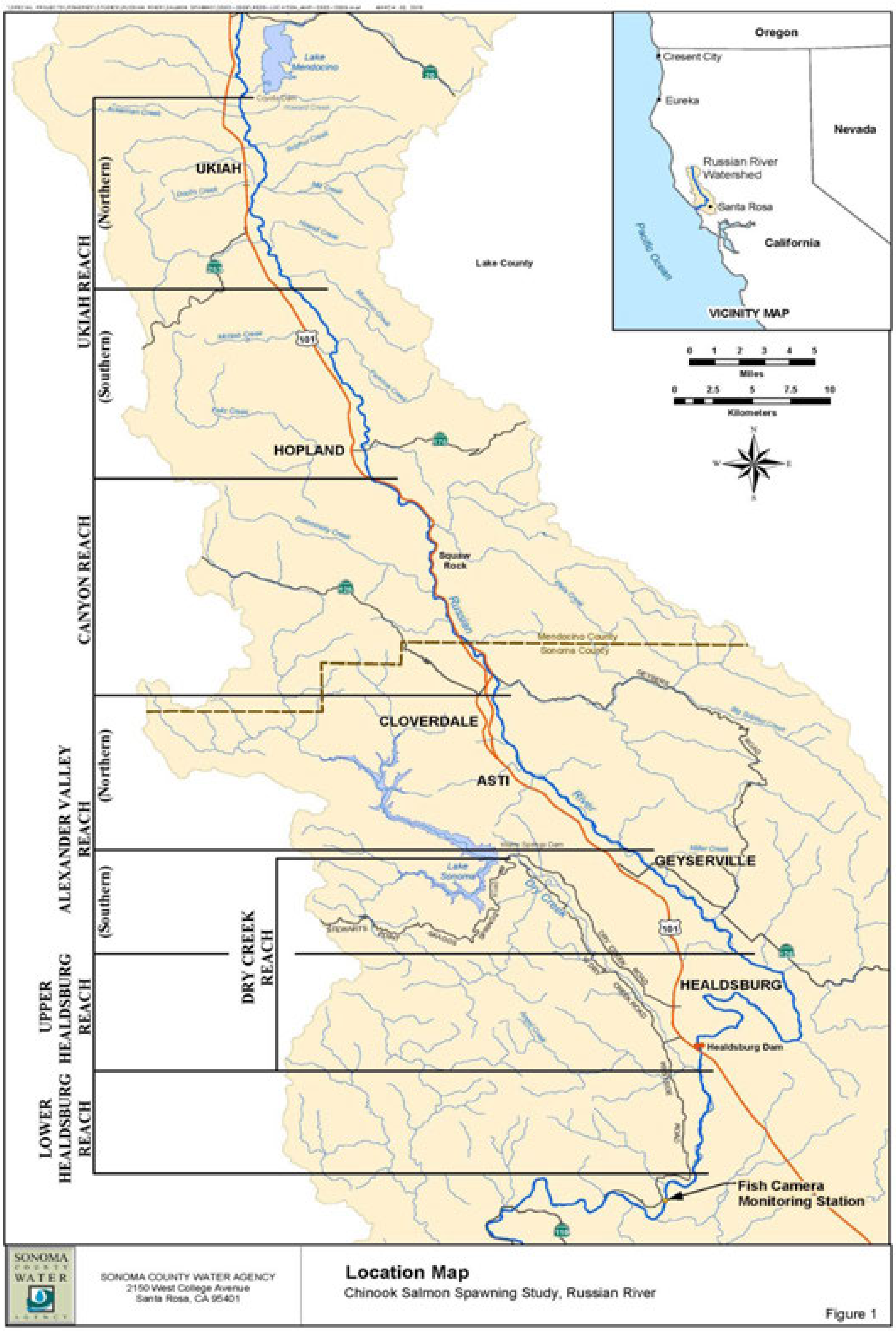

Figure 9.2. Chinook salmon spawning survey reaches. Only the Alexander Valley and Dry Creek reaches were surveyed in 2015.

Results

Most of the Chinook salmon spawning typically occurs in the upper Russian River mainstem and Dry Creek (Table 9.2). During 2015, there were 61 redds observed in the Alexander Valley reach of the Russian River, which is the lowest abundance recorded since surveys began in 2002. This survey was conducted prior to a large rain event that likely initiated additional spawning activity. Therefore, 61 redds is presumed to underestimate the actual number of redds produced in Alexander Valley reach. In Dry Creek, 215 redds were observed on December 16, 2015. This number is similar to the redd abundance recorded in several previous spawning runs (Table 9.2).

Conclusions and Recommendations

Although Chinook salmon surveys were restricted to two reaches in 2015 the distribution and abundance of redds appear to be similar to or within the range of other redd numbers observed during previous study years.

Table 9.2. Chinook salmon redd abundances by reach, upper Russian River and Dry Creek, 2002-2015. Redd counts are from a single pass survey conducted during the peak of fall spawning activity. *Survey either not completed or incomplete.

	Reach	Redd Observations (years)													
Reach	(rkm)	02	03	04	05	06	07	08	09	10	11	12	13	14	15
Ukiah	33.1	511	464	284	*	248	118	20	38	*	*	90[2]	81	*	*
Canyon	20.8	277	190	169	*	68	88	36	38	*	*	*	43	*	*
Alexander Valley	26.2	163	213	90	*	62	131	65	129	*	97	185	163	*	61[2]
Upper Healdsburg	25.6	79	40	8	*	23	67	48	38	*	66	53	57	*	*
Lower Healdsburg	8.2	6	0	7	*	1	2	9	30	*	7	4	18	*	*
Russian River	113.9	1036	907	558	*	402	406	178	273	*	170	332	362	*	*
Dry Creek	21.7	*	256	342	*	201	231	65[1]	223	269	229	362	325	130	215
Total	135.6	*	1163	900	*	603	637	243	496	*	*	*	*	*	*
Relative Contribution of Redds															
Russian River (%)	84.0	*	78.0	62.0	*	66.7	63.7	73.3	55.0	*	*	*	52.7	*	*
Dry Creek (%)	16.0	*	22.0	38.0	*	33.3	36.3	26.7	45.0	*	*	*	47.3	*	*

[1]Redd numbers are an estimate.

[2]Redd numbers are presumably an underestimate due to poor survey conditions.

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CHAPTER 10 : Synthesis

Introduction

The Sonoma County Water Agency has collected a variety of fish and water quality monitoring data relevant to fulfilling the overall monitoring objectives in the Reasonable and Prudent Alternative (RPA) of the Russian River Biological Opinion. Those efforts have been detailed in portions of this report leading to this chapter. The objectives specific to this synthesis chapter are to relate these data sets to one another first by illustrating the spatial and temporal extent of monitoring activities in the basin and second by presenting and discussing emerging trends in salmonid abundance, movement and growth in streams encompassed by the RPA.

As in previous years of RPA implementation, we collected fish and related environmental data from a broad spatial and temporal extent in the Russian River Basin (Figure 1). We collected juvenile and smolt data from multiple locations in Dry Creek, Mark West Creek, Dutch Bill Creek, Austin Creek and the Russian River estuary. We counted adult salmonids with an underwater video system on mainstem Russian River at the Healdsburg dam, a DIDSON system on mainstem Dry Creek at the mouth and we conducted seven repeat Chinook spawner surveys on the 22 km of stream length in mainstem Dry Creek downstream of Warm Springs Dam. Juvenile salmonds were sampled throughout the Russian River watershed using a variety of techniques. In the mainstem of the Russian River juvenile salmonids were sampled using beach seining at 10 fixed locations in the estuary and passive integrative transponder (PIT) antenna arrays operated near the upstream extent of the tidal portion of the estuary in Duncans Mills and adjacent to the golf course in Northwood and at points near the upstream extent of the river impounded by the Mirabel dam (Syar). Because of ongoing construction of a new fish ladder at the Water Agency's inflatable dam in Mirabel, neither downstream migrant trapping nor adult video monitoring could be conducted on the mainstem in 2015. In tributaries of the lower river juvenile salmonids were sampled using downstream migrant trapping with rotary screw traps on Mark West Creek at Trenton-Healdsburg Road and Austin Creek at the gravel mine as well as a funnel net on Dutch Bill Creek in Monte Rio. PIT antennas were operated in conjunction with downstream migrant trap sites on Austin Creek and Dutch Bill Creek. In Dry Creek juvenile salmonids were sampled using downstream migrant trapping with a rotary screw trap and backpack electrofishing. PIT antennas were operated in conjunction with the downstream migrant trap and additional PIT antennas were operated in main-channel and off-channel sites in Dry Creek. Complementary data on water quality were collected by means of continuously-recording data sondes at multiple sites throughout the estuary/lagoon and from bi-weekly and weekly grab samples at additional sites. Details regarding the specifics of water quality and fisheries monitoring activities are covered in individual chapters of this report.

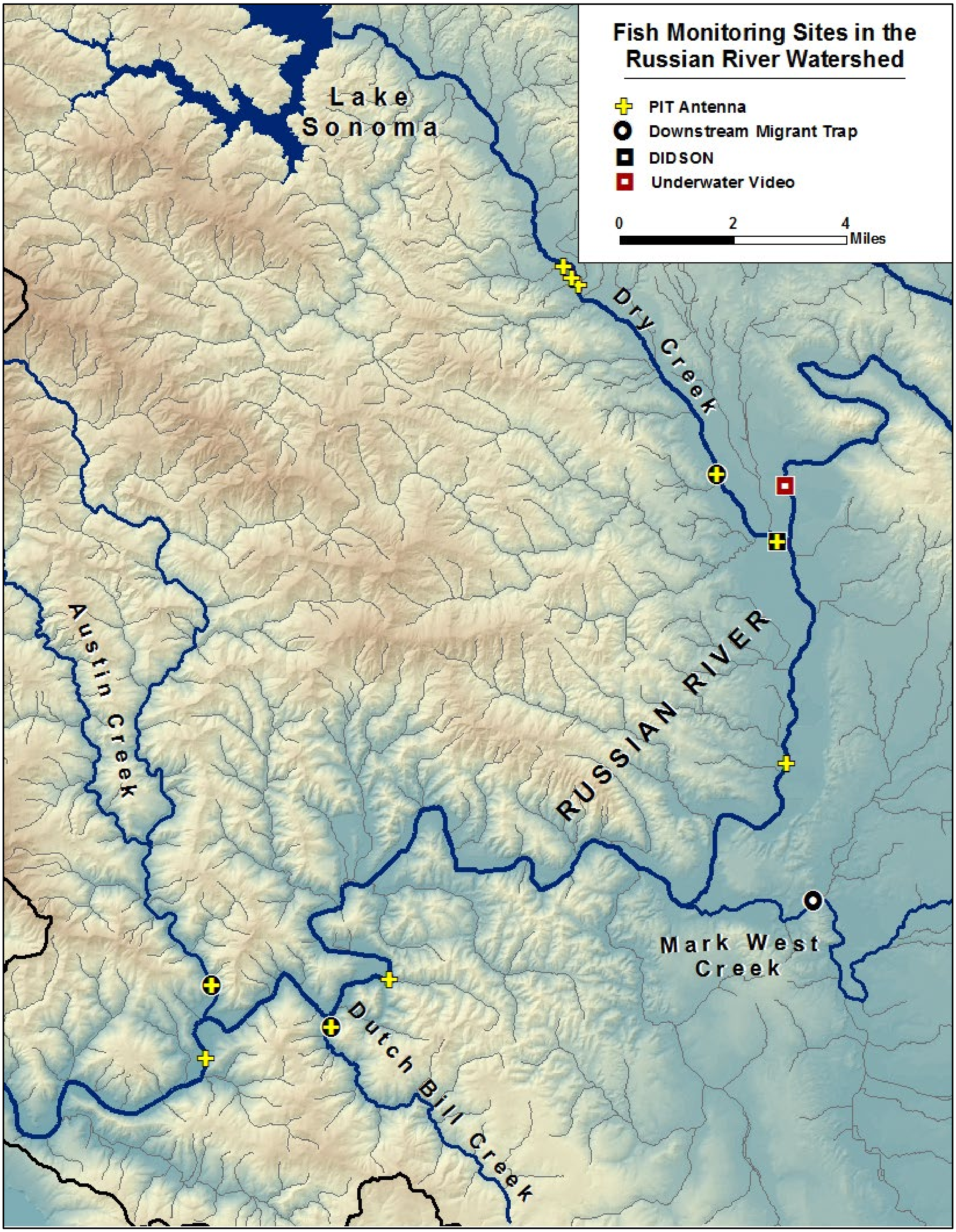

Figure 10.1. Spatial extent of fisheries monitoring related to the Russian River Biological Opinion, 2015. PIT antenna and downstream migrant trapping sites operated by UC Cooperative Extension/California Sea Grant are not shown.

In the sections that follow, we summarize population and movement dynamics of juvenile and smolt salmonids based on data from tributary and mainstem sites sampled in 2015. The Water Agency used PIT tags and fin-clipping as primary tools for characterizing population attributes. As described in other sections of this report and reports from prior years, PIT-tagged fish were detected during beach seining sampling in the estuary and at downstream migrant traps and stationary PIT-tag antennas located throughout the system (Figure 10.1). In the first section below, we broadly summarize available abundance information to describe some general temporal trends and variability in abundance. Following that, we focus specifically on movement of juvenile steelhead and Chinook salmon smolts from Dry Creek through the lower mainstem Russian River and estuary. We conclude with a discussion of the importance of consistent, broad-scale approaches to monitoring so that the effects of management on salmonid populations can be decoupled from environmental effects.

Abundance

Combined juvenile steelhead downstream migrant trap (DSMT) catch at Dry Creek, Dutch Bill Creek and Austin Creek was significantly lower in 2015 as compared to previous years. The decrease was most pronounced for Austin Creek (Figure 10.2). Juvenile steelhead density (from backpack electrofishing on mainstem Dry Creek) also showed decreases relative to recent years (Figure 10.3) and Chinook salmon smolt estimates in Dry Creek was lowest since trapping began in 2009 (Figure 10.3). Due to construction of a new fish ladder and fish screens at Mirabel, the Mirabel smolt trap was not operated in 2015. Captures of wild coho smolts were low everywhere (Figure 10.3). Relative to 2014, adult returns increased for Chinook and steelhead but decreased coho (Figure 10.4).

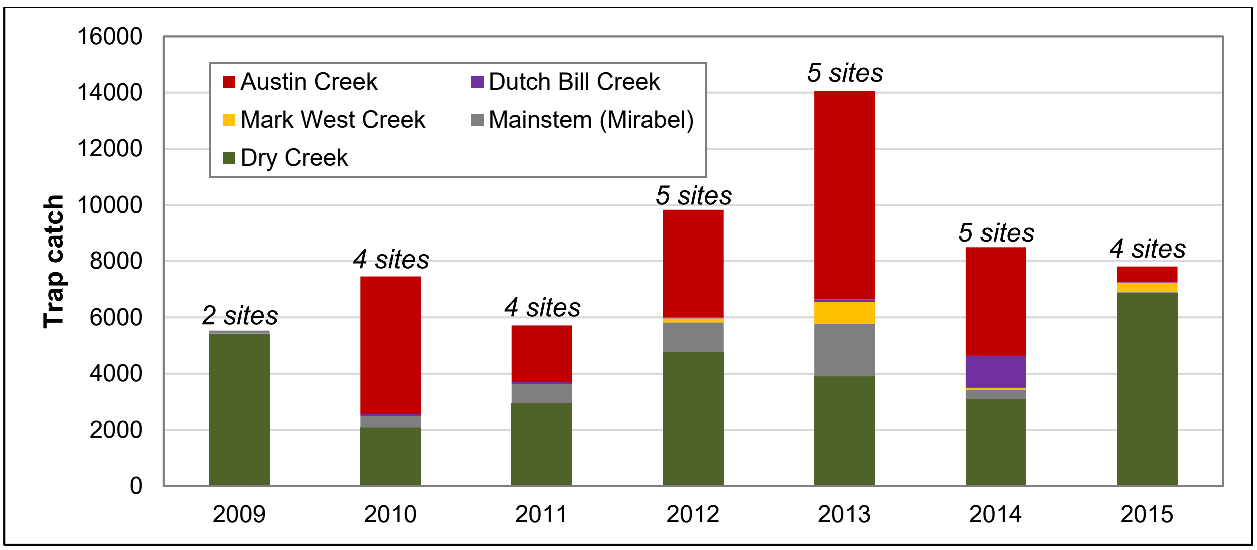

Figure 10.2. Number of juvenile (YOY + smolt combined) steelhead captured at downstream migrant trap sites operated by the Water Agency, 2009-2015 Note that downstream migrant trapping on the mainstem at Mirabel dam was suspended in 2015 due to construction of a new fish ladder.

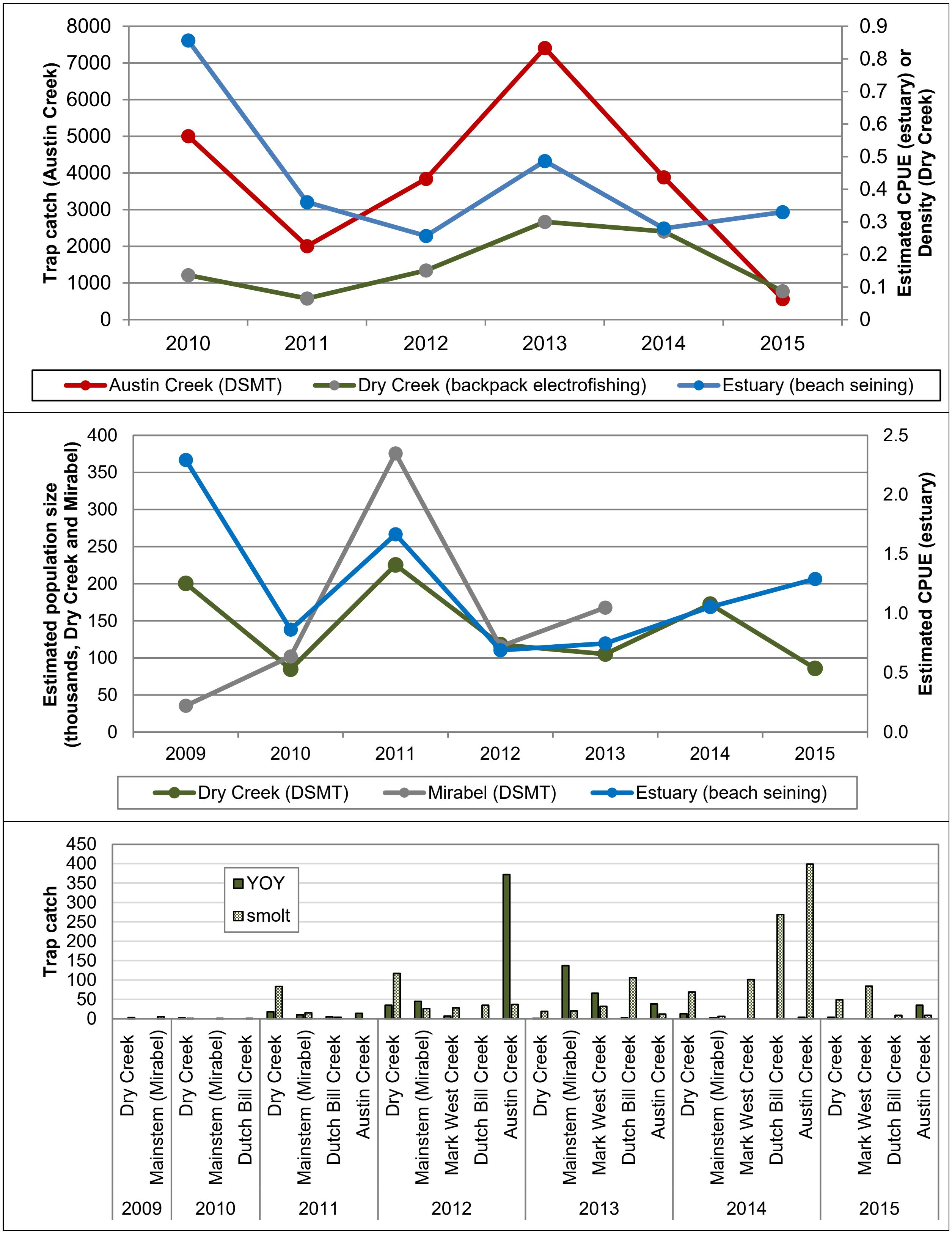

Figure 10.3. Indicators of juvenile steelhead (top panel), Chinook smolts (middle panel) and wild coho smolt/YOY (lower panel) trends based on monitoring conducted by the Water Agency, 2009-2015.

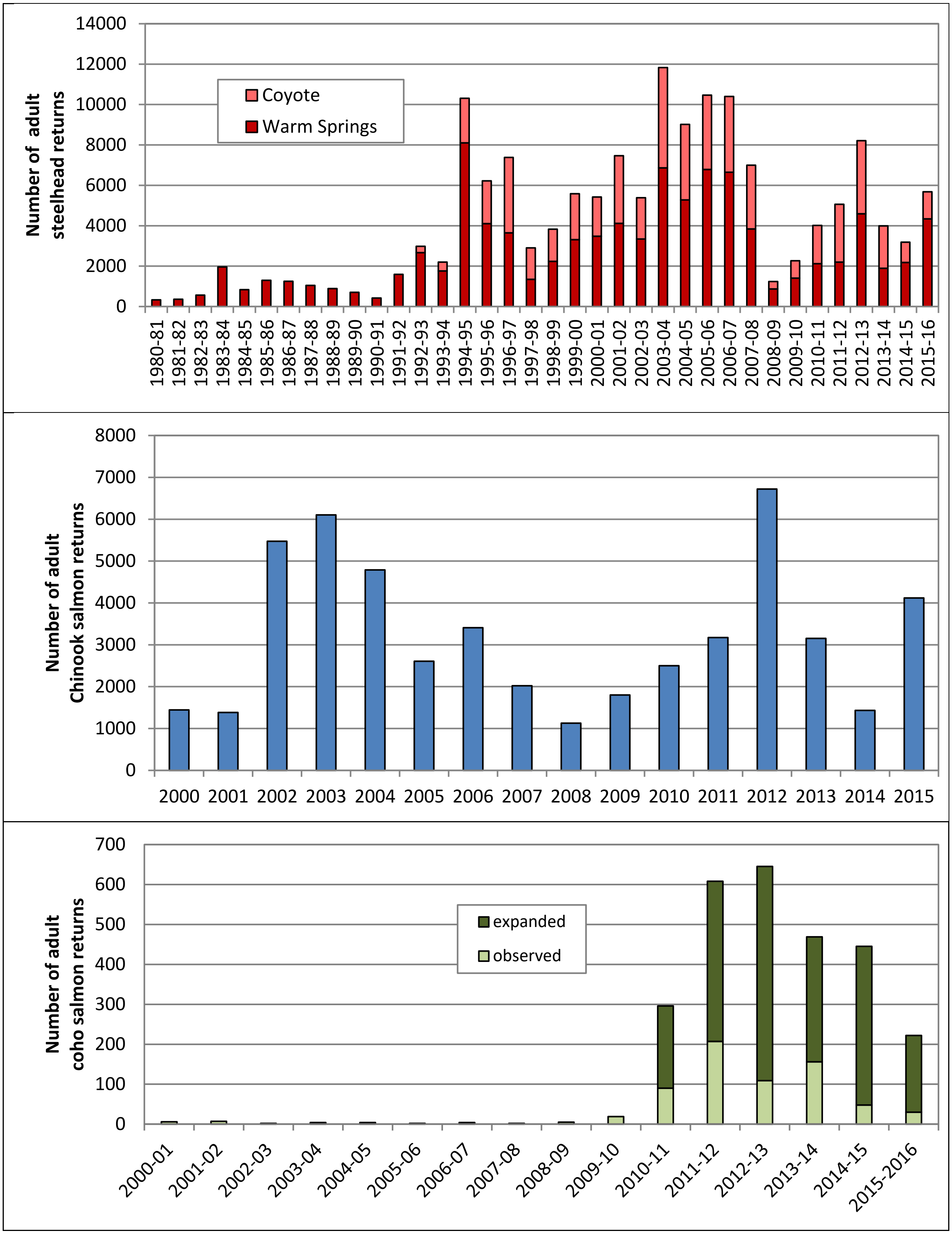

Figure 10.4. Indicators of adult steelhead (counted at Russian River hatcheries), adult Chinook (based on video-DIDSON counts at Wohler-Mirabel) and coho salmon returns (UC/CA Sea Grant).

Juvenile Steelhead and Chinook Smolt Movement

In 2015 we continued our evaluation of juvenile steelhead and Chinook smolt movement through the lower ~64 km of mainstem Russian. Unfortunately, because of construction at the Mirabel fish trapping site, we were unable to continue efforts to evaluate migration mortality of Chinook smolts as we have in the past (i.e., Manning and Martini-Lamb 2013). Our capabilities to make assessments of movement, however, was enhanced by operation of downstream migrant trap at Mirabel and PIT antenna arrays downstream of the Dry Creek trap site. When PIT-tagged fish left Dry Creek they could potentially be detected on a PIT antenna array near the mouth (rkm ~0.4), detected on the mainstem PIT antenna array near the upstream extent of the Wohler headpond (river km ~45) detected on the mainstem PIT antenna array near the community of Northwood (rkm ~19.5), detected on the PIT antenna array near the upstream extent of the estuary in Duncans Mills (rkm 10.5) or captured at during beach seining samples in the estuary (Figure 10.1).

In 2015, we only PIT-tagged 133 individual juvenile steelhead at all downstream migrant traps, combined. This was primarily due to the low fish captures at the Austin Creek trap resulting from the drought-shortened trapping season. We PIT-tagged an additional 87 juvenile steelhead during beach seining in the estuary and 1,671 while backpack electrofishing in mainstem Dry Creek (Table 10.1). We also PIT-tagged 1,367 Chinook salmon smolts at the Dry Creek trap (Table 10.2).

Despite the low numbers of juvenile steelhead captured at the Austin Creek trap, a significant number of those fish were detected leaving Austin Creek as evidenced by the high proportion of fish PIT-tagged at the Austin trap that were subsequently detected on the PIT antenna arrays near the mouth of Austin Creek and at the upstream extent of the estuary in Duncans Mills (75%, Figure 10.5). Movement rates out of the tributary of origin were fast (typically 1 day or less), but we did not see any evidence of that movement rate was related to either date or size at tagging.

Chinook salmon smolts typically moved through the ~3 km from the Dry Creek trap to the mouth of Dry Creek and the ~7 km from the mouth of Dry Creek to Syar on the mainstem Russian River in less than one day while the tme to travel to the estuary in Duncans Mills was approximately 3.5 days. There was some evidence that fish PIT-tagged later in the season moved through the system faster than fish tagged earlier in the season (Figure 10.6).

Table 10.1. Number of juvenile steelhead that were PIT-tagged and observed with a PIT tag at all Water Agency fish capture sites, 2009-2015.

Tributary	Survey	Year	Applied	Observed
Dry Creek	Downstream migrant trap	2009	0	2
		2010	9	2
		2011	0	3
		2012	0	2
		2013	2,704	59
		2014	1,354	36
		2015	0	3
	Backpack electrofishing	2009	688	94
		2010	789	158
		2011	708	112
		2012	763	202
		2013	694	143
		2014	1,060	168
		2015	1,671	237
Mainstem	Downstream migrant trap	2009	17	0
		2010	96	51
		2011	99	1
		2012	315	3
		2013	501	37
		2014	102	7
		2015	not fished	
Mark West Creek	Downstream migrant trap	2012	43	0
		2013	135	11
		2014	18	0
		2015	19	1
Dutch Bill Creek	Downstream migrant trap	2010	46	0
		2011	23	1
		2012	6	0
		2013	12	0
		2014	21	0
		2015	7	0
Austin Creek	Downstream migrant trap	2010	997	113
		2011	500	30
		2012	1,639	568
		2013	1,749	10
		2014	590	23
		2015	107	1

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Tributary	Survey	Year	Applied	Observed
Estuary	Beach seining	2009	68	4
		2010	241	41
		2011	88	18
		2012	85	15
		2013	43	4
		2014	174	29
		2015	87	2
		Total	***18,274***	***2,193***

Table 10.2. Number of Chinook salmon smolts that were PIT-tagged and observed with a PIT tag at all Water Agency fish capture sites, 2011-2015.

Tributary	Survey	Year	Applied	Observed
Dry Creek	Downstream migrant trap	2011	1,847	242
		2012	1,326	110
		2013	3,671	439
		2014	4,786	641
		2015	1,367	278
Mainstem	Downstream migrant trap	2011	0	45
		2012	0	36
		2013	0	202
		2014	777	256
		2015	not fished	
Estuary	Beach seining	2011	0	1
		2012	0	4
		2013	0	4
		2014	0	7
		2015	0	3
		Total	***13,774***	***2,268***

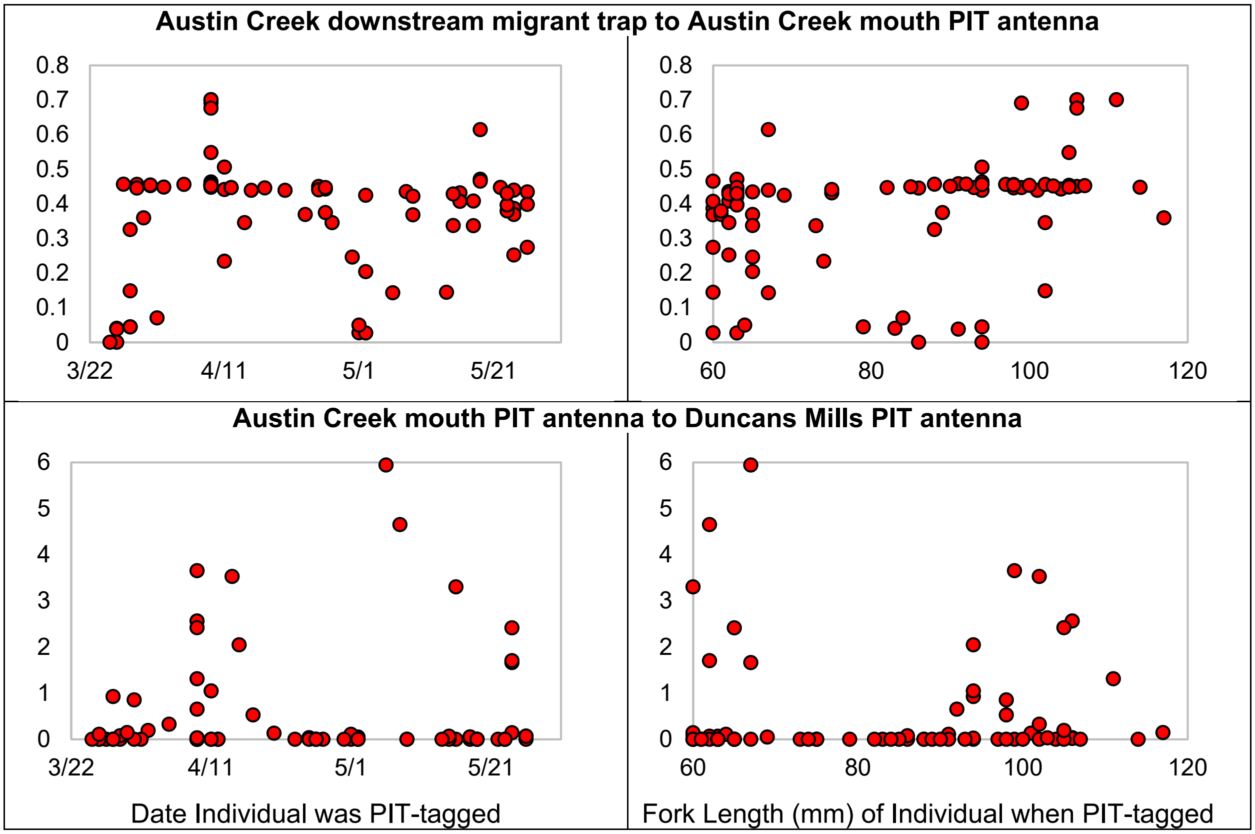


Figure 10.5. Movement rate (km per day) of individual juvenile steelhead PIT-tagged at the Austin Creek downstream migrant trap. Left-hand plots show movement rate as a function of date PIT-tagged and right-hand panels show movement rates as a function of size when PIT tagged.

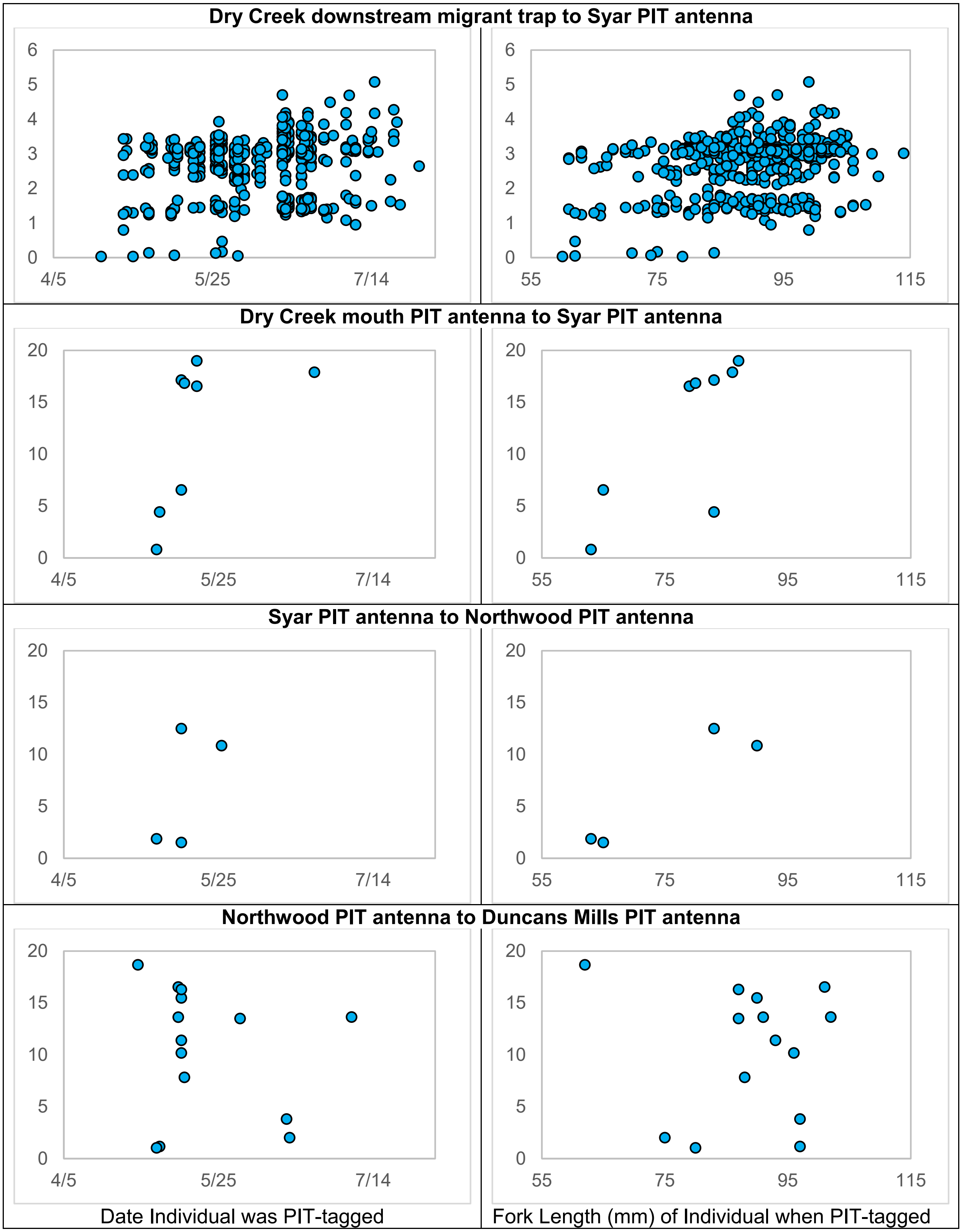


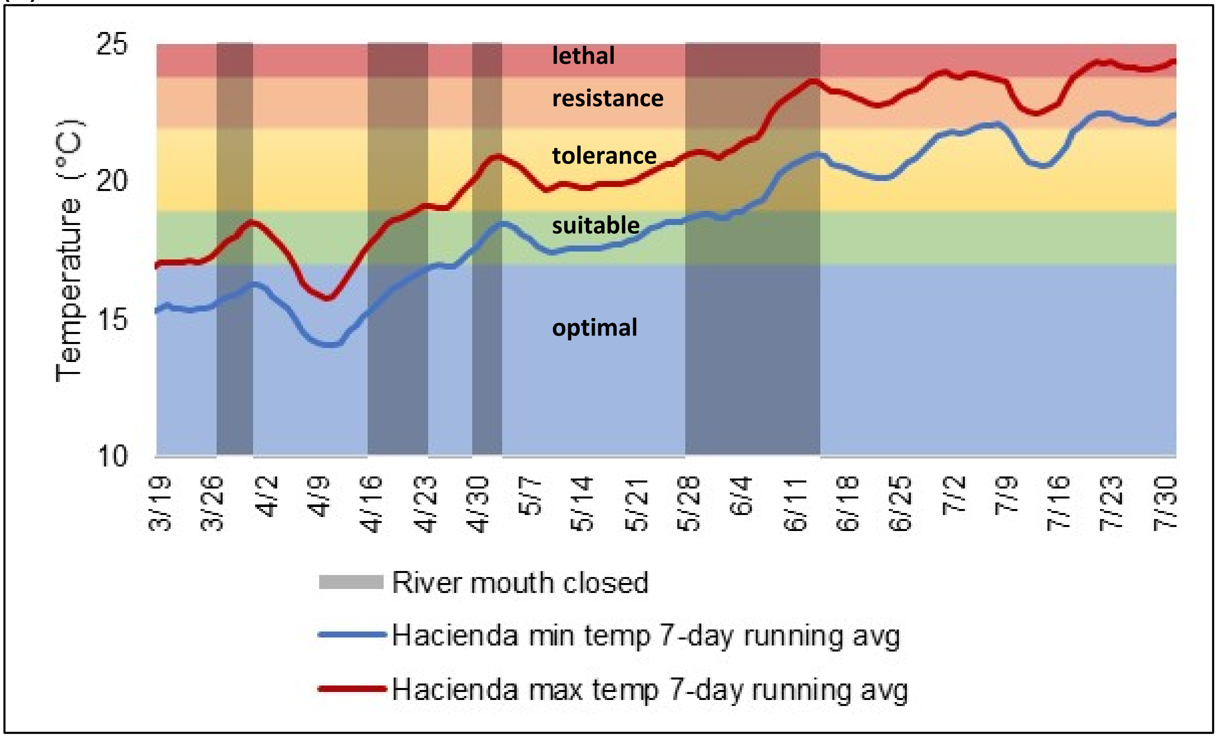
Figure 10.6. Movement rate (km per day) of individual Chinook smolts PIT-tagged at the Dry Creek downstream migrant trap. Left-hand plots show movement rate as a function of date PIT-tagged and right-hand panels show movement rates as a function of size when PIT tagged.

Conclusions and Recommendations

In 2015, the Water Agency continued to implement methods that will serve our need to understand the context in which salmon and steelhead populations in the Russian River are being affected by Water Agency actions as opposed to natural conditions that are simultaneously acting to shape these same populations. Continuation of California Coastal Monitoring Program (Adams et al. 2011) implementation throughout the watershed begun in 2013 by the Water Agency and UCCE should assist in providing a broader context in which to make those assessments.

2015 highlights the importance of understanding the abiotic factors that are recognized in the Russian River Biological Opinion as being important yet outside of Water Agency control. High water temperatures (Figure 10.7) occurred early in the year and likely impacted Chinook survival and perhaps juvenile steelhead more so than in any other year since we began implementing the RPA. Because of continuing drought conditions in 2015, fish not only faced high water temperatures but migrating coho and Chinook smolts were potentially exposed to poor water quality conditions for a longer duration. In addition, CMP monitoring revealed that, at least in in some tributaries, more than 70% of juvenile coho and steelhead observed during summer snorkel surveys were in pools that had dried up by the end of the summer. Without the combined efforts of multiple entities conducting fisheries monitoring throughout the watershed, this type of information would not be available thereby hindering our ability to make accurate assessements of project success.

(A) Juvenile steelhead



(B) Chinook salmon smolts

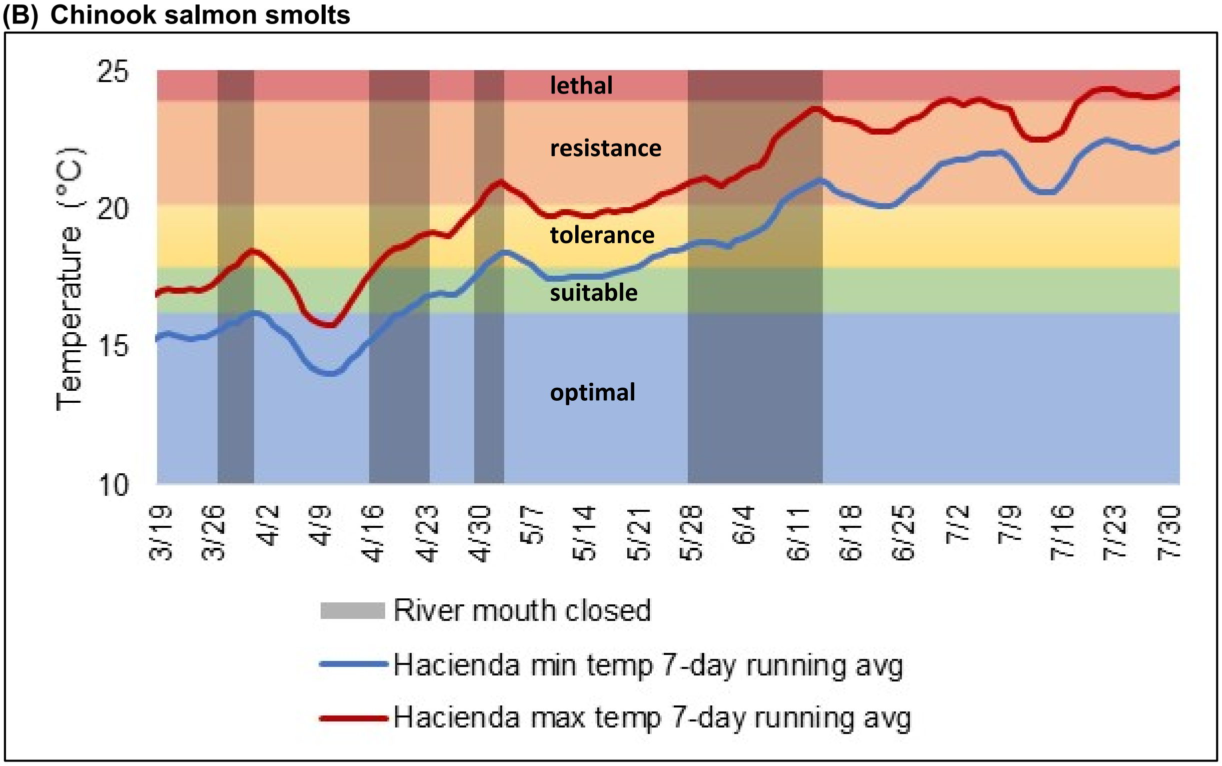


Figure 10.7. Water temperature at Hacienda (USGS gage number 11467000) and mouth closure periods during the period juvenile steelhead were PIT-tagged and released from Austin Creek (upper panel) and Chinook smolts were PIT-tagged and released from Dry Creek (lower panel). Temperature bins are from Sonoma County Water Agency (2016).

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